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Review of Dispersion Modelling for Odour Predictions

Science Report: SC030170/SR3

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Authors:
Dr Jon Pullen, Dr Yasmin Vawda

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Research Contractor:
Dr Yasmin Vawda Tel: 0207 902 6166
Bureau Veritas
Great Guildford House
30 Great Guildford Street
London
SE1 0ES

Environment Agency's Project Manager:
Neil Heptinstall, AQMAU, Cambria House

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Executive Summary

The Environment Agency's draft report *Horizontal Guidance for Odour*, H4 (Environment Agency 2002a) sets out an approach for quantifying odour annoyance, using a series of phases. These are estimation of odour release value, dispersion modelling to estimate the odour exposure, correlation of the predicted exposure against the expected degree of annoyance (using Indicative Odour Exposure Standards, IOES) and correlation with negative coping behaviours (nuisance and complaint). There are uncertainties associated with each step of this assessment process. This study has examined the component uncertainties by means of a detailed literature review, and identified their relative significance. An attempt is made at ranking the component uncertainties where possible using hypothetical, simplified scenarios.

This desk-top study has shown that not all of the information is available for quantifying the component uncertainties. However, this review provides a "model" of how the component uncertainties for each phase are inter-related. This could enable the Environment Agency to slot-in missing information as it becomes available from future research (or use professional judgement/consensus in the mean-time). In this sense, this report does provide a step forward in the practical understanding of the importance of the different phases of work that comprise an H4 assessment.

Source strength

The dispersion model requires the odour source strength as a key input. The most reliable value for this input could, in theory, be determined from a large number of periodic dynamic dilution olfactometric (DDO) measurements, on a single existing chimney, where the release is controlled, continuous, and does not vary with time or process cycle. The uncertainty escalates sharply for estimated odour emission rates, time-varying emissions, multiple sources on a site, and when specific compounds are used as surrogates for the total odour.

DDO is not currently practical on a continuous basis for any source. The inability to accurately quantify the odour's temporal variation, and difficulties in correlating the source variation with time-varying meteorology in the dispersion modelling, is the most significant source of uncertainty in the majority of odour assessments.

There are many components in the derivation of the source strength value for which a numerical estimate of uncertainty cannot be quoted. These are very situation-specific. The uncertainty on the source strength value can be several orders of magnitude even for commonly-encountered situations with time-varying emissions and/or estimates based on surrogate compounds or emission factors.

Dispersion Modelling of Exposure

It is important to recognise that the uncertainties associated with modelling some types of odorous release (e.g. diffuse/fugitive area sources, non-vertical vents) are very large. In such cases, the use of dispersion modelling as an assessment tool should be questioned.

The results of different, new-generation models can vary by up to a factor of 8, for high percentile calculations with significant building wake effects.

Examination of the *range* of results provides a sensitivity analysis of the model algorithms, and provides greater confidence in any regulatory decision.

Dispersion modelling is usually carried out when the risk of odour annoyance is high. Under these circumstances, the use of more than one dispersion model can be justified for a risk-based approach.

There is an urgent need to verify, for UK situations, the Dutch dose-response relationship which was established historically for livestock units. This is because almost no details are available on the dispersion model which was used to establish this empirical dose-response curve, nor the input data for that modelling. Of particular concern is the reliability of the source strength data that were used for the Dutch modelling.

There are many components of the dispersion modelling process for which a numerical estimate of 'typical' uncertainty cannot be quoted. These are situation-specific, e.g. when complex terrain is present.

Dispersion models are currently in practical use only for predicting 'ensemble mean' (typically hourly mean) concentrations. Fluctuation modelling is not yet adequately validated. As long as this remains the case, the 'Type 2' approach for odour assessment set out in Draft H4 (hourly mean modelling compared against an empirical benchmark) must remain the only feasible option.

ADMLC has provided detailed guidance on best practice for dispersion modelling. Most of the guidelines are applicable to odour modelling. It would be useful if more detailed guidance was incorporated into Draft H4, giving a clearer steer towards the required level of transparency and rigour in odour assessments. Also, some uniformity in the way that model sensitivity and uncertainty are expressed would be useful.

Correlation with annoyance

The uncertainty in the correlation of odour exposure with odour annoyance is a combination of two types of component uncertainties: random (which are precision-type uncertainties, responsible for the scatter and the correlation coefficient found for the dose-response curve), and systematic (which are the bias or accuracy-type uncertainties that include factors such as how relevant the Dutch pig-odour response curve is to other odour types in UK conditions, and how appropriate the concentration levels for the 'offensiveness' bands have been set).

If a series of dose-response studies had been carried out under UK conditions, it would have allowed the repeatability of the Draft H4 method to be estimated. Unfortunately no such studies have been carried out.

The use of a calibration curve derived from Dutch livestock odours, and applying it to other odours and other types of installation in different countries, presents an additional layer of uncertainty compared to deriving a modelling guideline from a bespoke, dose-response study. However, again it is not possible to place any numerical estimates of the magnitude of this additional layer of uncertainty, as it is situation-specific. For a livestock installation the additional layer of uncertainty might be expected to be small. For other, very different, odours or installations, the leap of faith is wider and the additional uncertainty may be much larger. Practitioners must make themselves aware of this, and form a qualitative view on the significance of this component of uncertainty.

On the positive side, the level of annoyance measured by a survey in New Zealand (Ministry for the Environment 2002) was found to be consistent with the odour dose—community-response curves reported for the Netherlands (Miedema 1992). The dose-response curves, although developed for other industries and using a Dutch community response, appeared to be valid for pulp mill odours in New Zealand.

The Draft H4 guidance states that the above benchmarks are indicative standards and that UK dose-effect studies are planned. It also states elsewhere in the document that *“the only realistic way of estimating the actual level of annoyance in a particular community resulting from exposure is by carrying out dose-response studies locally”*. However, Draft H4 appears much less explicit than the New Zealand guidance in highlighting the “interim” nature of these generic-type odour guidelines and that they should ideally be superseded by industry-specific guidelines developed from bespoke dose-response studies.

It is possible that some dose-response studies will be performed around waste management facilities as part of a study into defining loss of amenity through odour, carried out as part of Defra’s Waste Research R&D programme*. There is also a possibility of UK Water Industry Research (UKWIR) coordinating some studies around wastewater treatment plants to support the water industry in meeting the Defra *Code of Practice on Odour Nuisance from Sewage Treatment Works*.

Correlation with nuisance and complaints

The Environmental Protection Act (EPA) 1990 contains no technical definitions of nuisance, such as maximum concentrations, frequencies or durations of odour in air. This needs to be addressed, together with a nuisance measurement methodology, before any estimate of the uncertainty in the correlation with odour exposure can be made.

* Details at <http://www.defra.gov.uk/environment/waste/wip/research/index.htm>

Complaints are more usually measured directly by complaints monitoring, rather than being predicted. However, dose-response studies using complaints as the response measurand have been carried out in New Zealand and Australia, but using different models to those in common use in the UK, and using different percentiles to describe exposure.

The uncertainties in correlating predicted exposure with either nuisance or complaints levels would be considerably higher than for annoyance, due to the additional factors involved.

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1 Introduction

1.1 Scope and status of this report

This Project, *Review of dispersion modelling for odour predictions*, is one of a cluster of three projects funded by the Environment Agency's Science Department (reference 13933), managed by Bureau Veritas (BV). BV provided the Environment Agency with a proposal and Work Plan for Project 3 on 1st November 2005, in accordance with the Technical Specification dated 12th October 2005 from Dr Damien Rosser. The proposal committed BV to providing the Environment Agency with a report summarising the findings of an investigation into the uncertainties associated with using atmospheric dispersion modelling to assess odour exposure, and extrapolation to annoyance.

1.2 Background and overall aim of the project

In October 2002, the Environment Agency published its draft Technical Guidance Note H4 (Draft H4) (Environment Agency 2002a). This guidance described an approach to assessing and regulating odour impacts, involving quantifying odour emissions, dispersion modelling to estimate odour exposure, and correlation of exposure with the expected degree of annoyance using "*Indicative Odour Exposure Standards*" (numerical air quality criteria for odour). The approach can be used directly to assess the annoyance impact of an installation. Alternatively, it can be worked backwards from an "acceptable" level of annoyance to derive the maximum emission limit value (ELV) for odour at source that can be set as a Pollution Prevention and Control (PPC) Permit condition that will avoid "*reasonable cause for annoyance*".

The approach contains a number of more or less discrete steps, each having its own component uncertainty. It is important to be able to understand how important these component uncertainties are, which ones dominate, and which ones are relatively insignificant. This project aims to provide a better understanding of the likely *relative* importance of the various steps in the Draft H4 approach, in terms of their uncertainties. Ultimately, the findings of this project could assist in improving the Draft H4 method, by ensuring that effort in revisions is targeted at those uncertainties which are most important for permitting decisions.

1.3 Objectives of the Study

This review focuses on installations for which predictive dispersion modelling may be used, as part of a detailed odour impact assessment for PPC permitting. These may be existing installations with an existing or potential odour problem; or they may be proposed installations with potential odour problems. The drivers for this review of uncertainties are as follows:

- a) Appendix 3 of Draft H4 Part 1 states that a sensitivity analysis of model predictions for critical model input parameters should be carried out. Conclusions and assessment need to take into account uncertainties in model predictions. Both the operator and regulator need to have confidence in the outcome of the dispersion modelling.
- b) If modelling shows that the Draft H4 Indicative Odour Exposure Standards (IOES) are, or are not, met, there needs to be confidence in the certainty of this prediction. The uncertainties in the source strength and the modelling need to be identified.
- c) Also, there has to be confidence that the Indicative Odour Exposure Standards really do represent '*no reasonable cause for annoyance*' for the given situation. This requires a review of how the Indicative Odour Exposure Standards were derived, and how applicable they are to the situation being modelled. A review of the derivation the Indicative Odour Exposure Standards has been carried out in the Literature Review for Project 2 of this Science Project (Environment Agency 2007b).

1.4 Applications and limitations

1.4.1 Using the information on uncertainty

This study seeks to examine the uncertainty (U) in the main stages of an odour impact assessment that follows the Draft H4 approach. Particular attention is given to:

- the value for source strength used in the dispersion model (discussed at length in Chapter 4);
- the differences between the predictions of different models. These are important, because the Draft H4 Indicative Odour Exposure Standards were set on the basis of results from a particular type of dispersion modelling exercise (based on an 'old-generation' model) in the Netherlands (discussed at length in Chapter 5). Different (i.e. 'new-generation') models are now used in the UK and elsewhere for odour assessment.

The relative magnitudes of the component uncertainties (in the source strength, in the modelling results, and in the assessment criterion) need to be examined in order to prioritise future research and audit effort.

A realistic outcome of this review is not so much to get a precise estimate of the overall uncertainty, but rather to investigate the relative importance of the main component uncertainties. It should be possible to identify, for example, if the dispersion modelling has a much more significant component uncertainty than emissions quantification, or assignment to different bands of unpleasantness.

1.4.2 Source-types covered by this study

There are three main categories of industrial releases to atmosphere.

- **Controlled releases** – the emissions are managed in some way, either as part of a process or as part of a control/ abatement mechanism and the emissions are therefore quantifiable. Most (but not all) controlled releases are from point sources: BS EN 17025 defines a point source as a discrete stationary source of waste gases to atmosphere through canalised ducts of defined dimension and air flow rate (e.g. chimneys, vents).
- **Diffuse releases** - BS EN 17025 defines diffuse sources as those with defined dimensions (mostly surface sources) which do not have a defined waste-air flow, such as waste dumps, lagoons, fields after manure spreading, un-aerated compost heaps.
- **Fugitive releases** – these are, literally, releases that cannot be captured. BS EN 17025 defines fugitive sources as elusive or difficult to identify sources, releasing undefined quantities of odorants (e.g. valve and flange leakage, passive ventilation apertures). They are uncontrolled and often dependent on external conditions (e.g. wind) which make them difficult to quantify with any reasonable degree of certainty. Another definition of a fugitive emission is a release that is unintentional. An oil refinery may have a quarter of a million pumps, valves and flanges that potentially can leak, making it impractical to measure the emissions from every source.

Industrial releases to atmosphere can also be subdivided in terms of their spatial characteristics, usually as a point source, line source or area source. It is important to recognise that these can be controlled releases, diffuse releases or fugitive releases, as shown in the table below. However, it is controlled point sources (e.g. chimney stacks and vents) that are most commonly monitored and modelled.

Table 1.1 Categories of emissions to atmosphere

	Point source	Line source	Area source
Controlled release	Emissions from fixed-location plant and often (but not always) released to atmosphere via a vent, duct or chimney stack	Tail-pipe emissions from vehicles driving along a road	
Diffuse release	Open process tanks	Open gullies and culverts	Landfill surfaces, lagoons, compost heaps
Fugitive release	Intermittently leaking valve	Dust re-suspended in a vehicle's wake; wind-whipping of dusty material on an open conveyor belt	Wind-whipping of a stockpile of dusty material

The main focus of the discussion in this study is on controlled point-source releases of odour, because this is the main application of the Draft H4 modelling approach. The discussion is not designed to be focussed on diffuse or fugitive sources, as the Draft H4 approach, using modelling to back-calculate permissible ELVs, is not applicable to such situations. However, for the sake of completeness, some limited comments on other source types are included, as it is recognised that the Environment Agency will, from time to time, receive Environmental Impact Assessments (EIAs) that include odour impact predictions for diffuse (usually area) sources.

Emissions factors and mass balance calculations are applied most commonly to diffuse sources and fugitive sources that are difficult to monitor. For similar reasons, the Draft H4 approach using modelling to back-calculate permissible ELVs, is not applicable to emission factor and mass balance calculations, and only limited comments have been made on these.

2 Methodology

2.1 Overall approach to the investigation

It was agreed with the Environment Agency that this investigation would be a desk-top study, based upon a detailed literature review. No practical investigations have been carried out using, for example, sampling, analysis or dispersion modelling.

2.2 Literature review

The objectives of Project 3 were designed to be met by a literature review of the available and most up-to-date UK and international scientific literature pertaining to the steps comprising an odour impact assessment *which includes dispersion modelling*, with a view to assessing the uncertainties at every step.

Draft H4 was published by the Environment Agency in October 2002, using the best information and data which were available at the time. It is desirable for Project 3 to focus on relevant new work published *after* the issue of Draft H4. Relevant publications which are more recent than October 2002 have been identified via a 'Google' search. Copies of all documents are available to the Environment Agency from BV on request.

Several key publications contain information which was the most up-to-date at the time that they were published. However, certain findings can no longer be considered as entirely relevant. Notable as an example are papers which describe model inter-comparison studies; the versions of most models mentioned in the papers have inevitably been superseded (Hall *et al* 2000a). This is discussed in more detail in Chapter 5.

3 The Component Steps in the Annoyance Process

In examining the uncertainty (U) associated with a measurement or estimation process, the first task is to break it down into discrete parts to simplify the task. The component uncertainties of these stages can then be investigated and, if required, combined to give an estimate of the overall, total U.

Van Harreveld (2001) proposed a conceptual flowchart (which has been set out in previous Environment Agency (2000b) documents) showing the relationship between exposure to malodour and its potential effects on a human population; this is reproduced in Figure 3.1. The contributing factors and the effects, which may result ultimately in complaints, are far from straightforward and few of the relationships are completely understood. The main factors, discussed in detail in the Literature Review for Project 2, were summarised by Van Harreveld as:

- the characteristics of the odour that is released, i.e. detectability (odour concentration), intensity, hedonic tone, and annoyance potential;
- variable dilution in the atmosphere through turbulent dispersion (turbulence or stability of boundary layer, wind direction, wind speed, etc.);
- exposure of the receptors in the population (this will be dependent on the odour concentration in the ambient air and also the frequency and duration of episodes, influenced by the location of residences, movement of people, time spent outdoors, etc.);
- context of perception (e.g. other odours, background of odours, activity and state of mind within the perception context);
- receptor characteristics (exposure history, association with risks, activity during exposure episodes, and psychological factors such as tolerance and expectations of the exposed subjects, their coping behaviour, their perceived threats to their health).

Once exposure to odour has occurred, the process that may lead to annoyance, nuisance and possibly complaints, will involve many psychological and socio-economic factors. These factors, and technical definitions for the terms annoyance and nuisance, are described in the Literature Review for Project 2.

The Literature Review for Project 2 (Environment Agency 2007b) describes the factors that determine the impact of an odour: whether an odour has an objectionable effect depends not just on the strength of the odour, but also its frequency, intensity, duration, offensiveness (the character and unpleasantness – hedonic tone – at a particular intensity) and location of the receptors. These attributes, known collectively as the FIDOL factors, need to be incorporated into (or otherwise accounted for in) the numerical benchmark criterion.

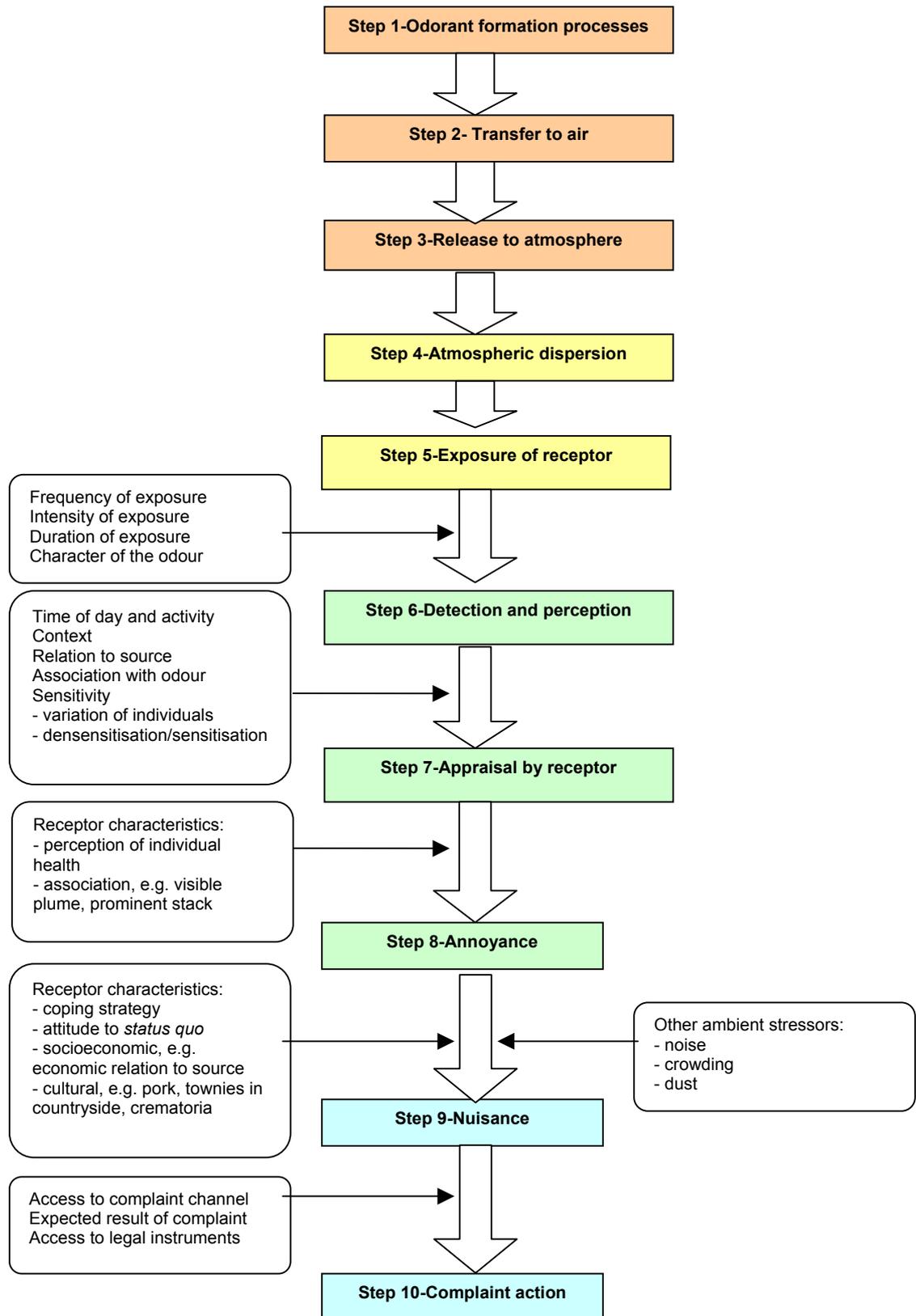


Figure 3.1 From odour formation to complaint, based on Van Harreveld (2001)

The Literature Review for Project 2 describes the two approaches that can be used to develop an air quality criterion for modelling/monitoring odour that incorporates these factors. The first ('Type 1') is based on a so-called "deterministic" theoretical approach that attempts to incorporate from first principles the FIDOL factors. However, the earlier Environment Agency Science Report (2007b) concluded that with the current level of understanding, such attempts were typically too simplistic to be effective and, as for noise, regulation of odours would be better served by a straightforward, practical approach, even if this did not necessarily involve all the concepts and refinements. The Environment Agency research favoured another type of numerical benchmark ('Type 2'), where odour guidelines are derived from the empirical relationship between odour exposure (measured or modelled, with the Environment Agency emphasis on modelled exposure) and annoyance (measured by a community survey), as shown in Figure 3.2. Numerical benchmark criteria are the foundation for assessing impact using predictive modelling.

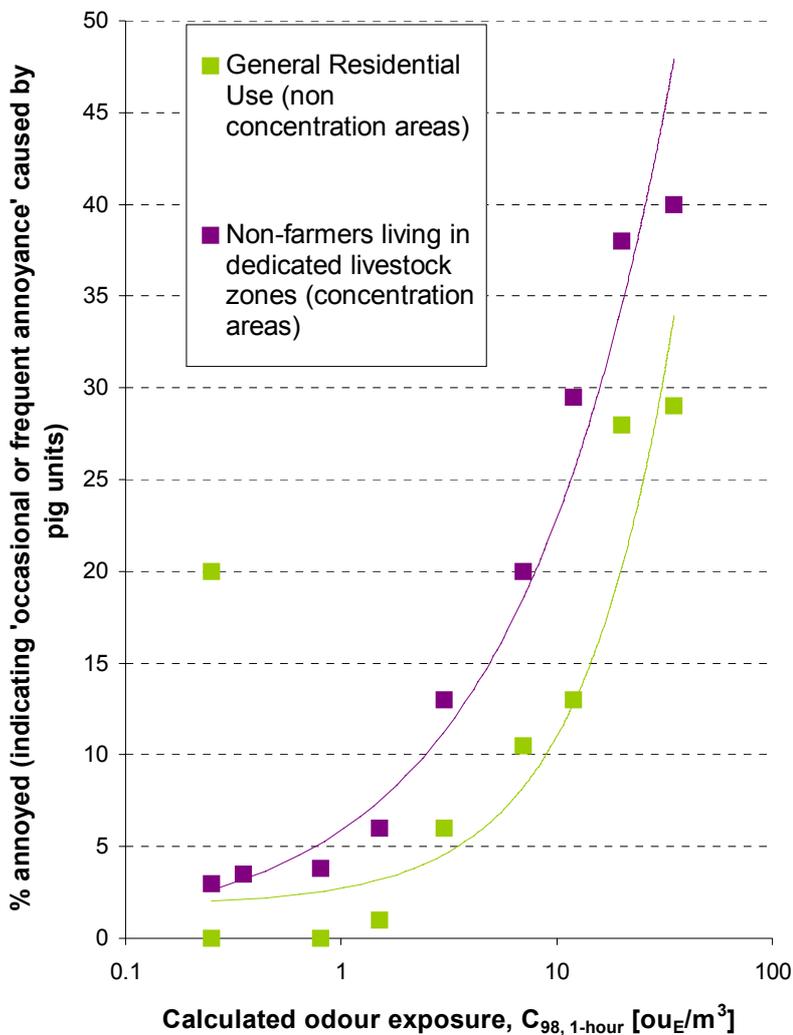


Figure 3.2. Dose-response relationship for livestock odours in The Netherlands (after Bongers *et al.* 2001)

This led to the Environment Agency developing its numerical benchmarks for odour mixtures that were put forward as “Indicative Odour Exposure Standards” in the Draft H4 guidance.

This “Type 2” approach, based on an empirical, epidemiological dose-response study, does not normally concern itself with the details of the FIDOL factors, and instead treats the process as a “black box”. The overall uncertainty of the Draft H4 method will be a combination of:

- *Random component uncertainties* – these are precision-type uncertainties, responsible for the scatter and the correlation coefficient found for the dose-response curve; and
- *Systematic component uncertainties* – these are the bias or accuracy-type uncertainties that include factors such as how relevant the Dutch pig-odour response curve is to other odour types in UK conditions, and how appropriately the concentration levels for the “offensiveness” bands have been set.

If a series of dose-response studies were carried out under UK conditions, it would allow the repeatability of the Draft H4 method to be estimated. However, the scope of this project is not to provide an experimental estimate of the overall uncertainty or the repeatability. It is to examine the main components of the Draft H4 method to determine which ones have an important effect on the overall uncertainty, and which ones less so. Figure 3.3 shows the main components of the Draft H4 assessment method, and how they encompass the steps shown previously in Figure 3.1.

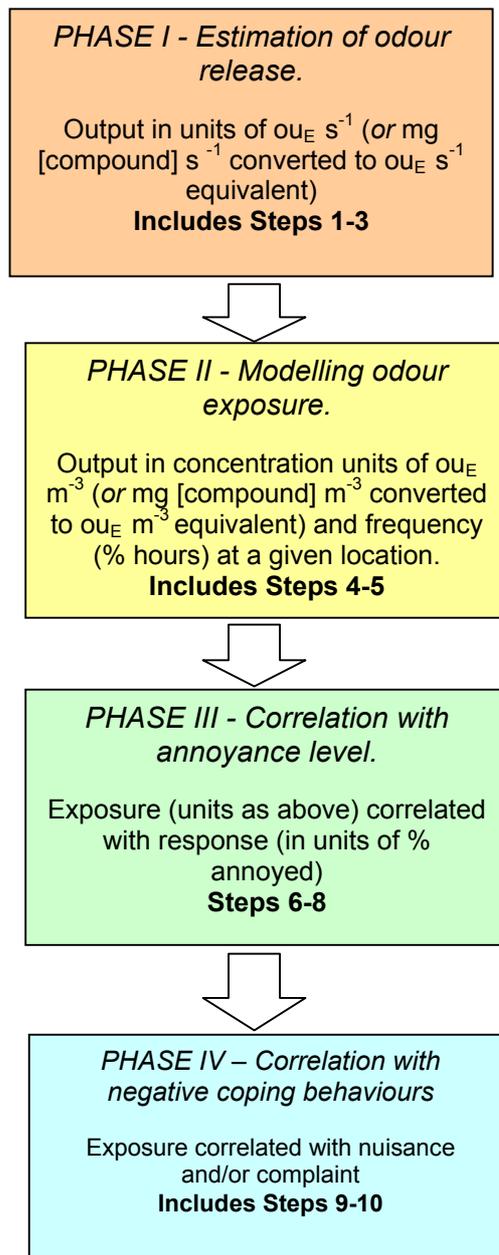


Figure 3.3. A summary of the phases involved in assessing odour impact using the Draft H4 methodology

It is important to be clear that atmospheric dispersion modelling of odorous emissions, on its own, merely allows the prediction of odour exposure (correlating with Step 5). The assessment process can only be extended further to estimate annoyance (correlating with Step 8) if that exposure prediction can be correlated with the level of odour annoyance. Draft H4 does this by requiring comparison with “*Indicative Odour Exposure Standards*”, based on an epidemiological dose-response relationship (where the measured response was 10% of the population annoyed).

It is currently beyond the scope of the Draft H4 methodology to extend the assessment process to correlation with such negative human coping behaviours as nuisance and complaint (Steps 9 and 10). This would require a dose-response relationship with the community response measured in terms of complaint action. The use of complaints records is discussed further in Chapter 7.

4 Phase I – Estimation of Odour Release Value

4.1 Overview of the odour release process

It is necessary to understand the processes involved in the odour release, to help assess how well we are able to estimate the magnitude of that release. The steps are outlined in Figure 4.1.

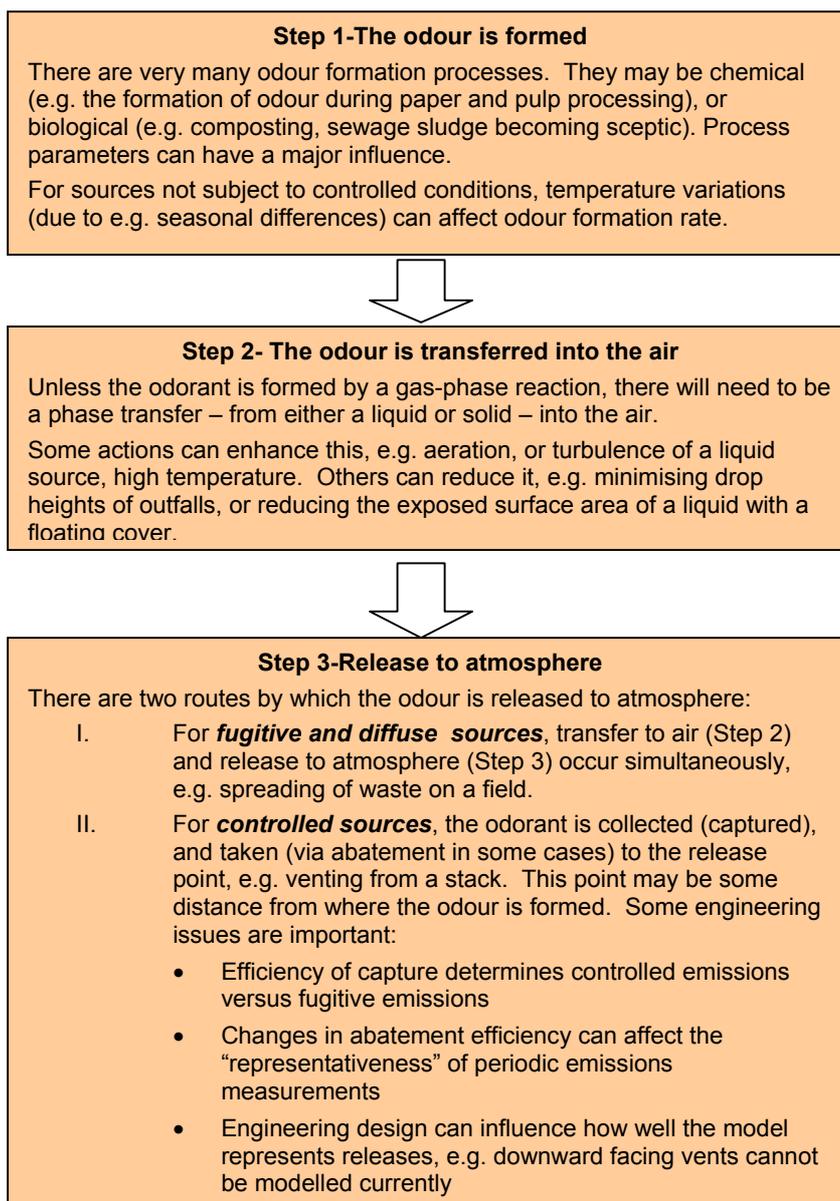


Figure 4.1 The steps involved in the odour release process

4.2 Summary of the component uncertainties in the odour release rate value

The component uncertainties that go towards making up the uncertainty associated with the odour release value (that is input into the dispersion model) are summarised in Figure 4.2, with further details on these individual uncertainty components given in Section 4.3.

It is crucial to understand that there are many different ways in which the odour release value can be obtained. Figure 4.2 attempts to show the choices available to the practitioner. One of the most basic choices is whether the odour release rate value will be obtained by measurement, or by estimation (see Box 4.1 for clarification of these terms). Both have their place – and are allowed in the Draft H4 methodology – but as is explained later in this chapter, the choices taken at this early stage can have big influences on the resulting uncertainty of the final odour release value that goes forward into the modelling phase of the assessment.

In looking at Figure 4.2, the first distinction that needs to be made is between:

- a) the uncertainties associated with the technique *itself* (whether that be measurement or estimation); and
- b) the uncertainties associated with the *application* of that technique to a particular scenario/situation.

The next distinction made is that between the two alternative routes to obtaining the odour release value, namely:

- I. measurement of the odour release directly: or
- II. estimation of the odour release.

For an existing installation there may be a choice on whether the release values will be based on measurements or estimations. If the installation is not yet in operation, the releases will need to be estimated. Whichever technique is used will have its own component uncertainties.

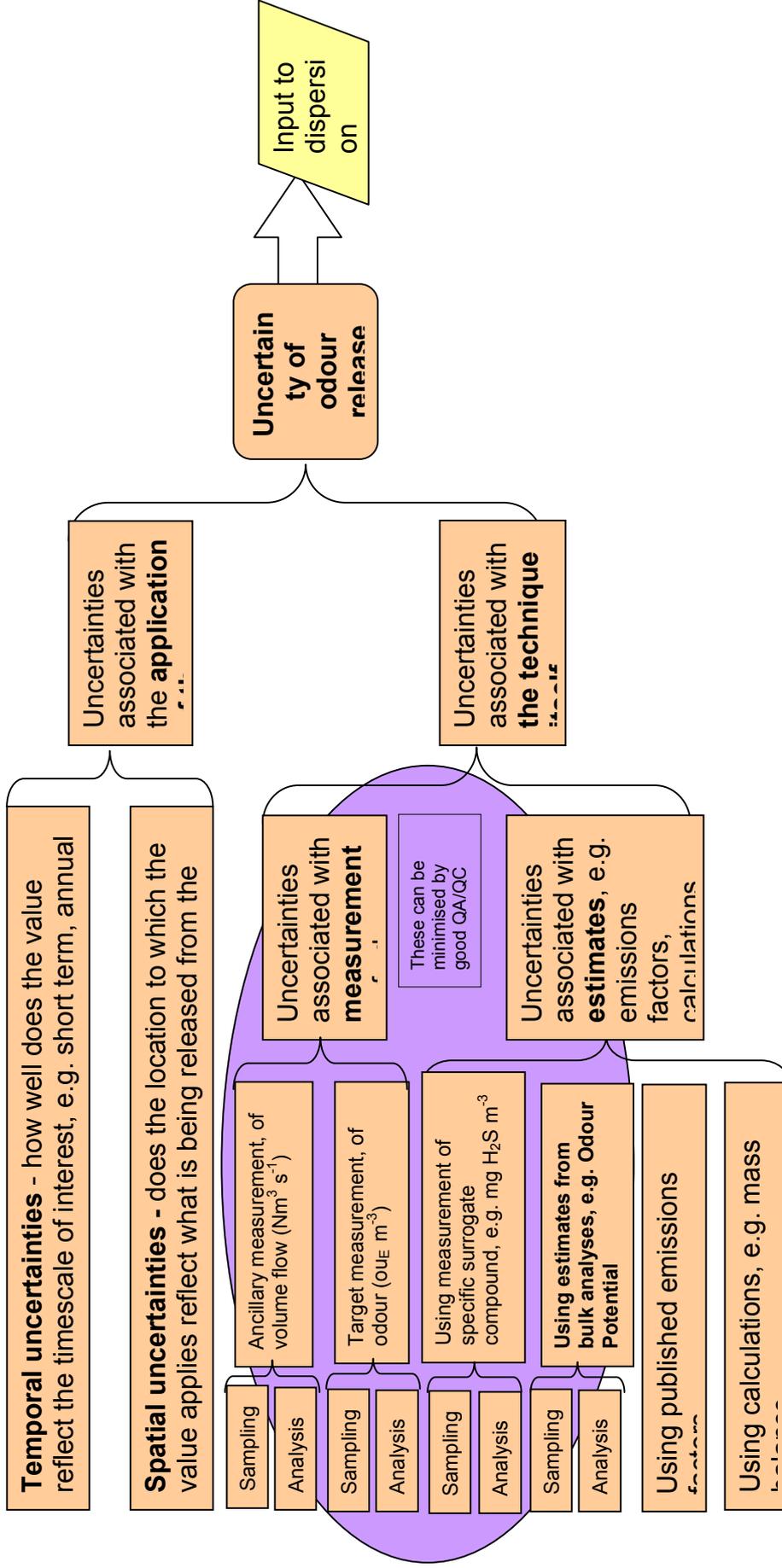


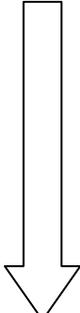
Figure 4.2 Summary of the component uncertainties in the odour release rate value

Box 4.1 Terminology – monitoring, measurement and estimates

The terms measurement and monitoring are often used interchangeably, but they have specific and different meanings. **Measurement** of an environmental parameter involves assigning a numerical quantity to its magnitude by carrying out some kind of gauging. **Estimation**, on the other hand, is the approximate judgement or opinion of this magnitude. **Monitoring** involves collecting and interpreting a number of measurements or estimates over a period of time. It usually involves an element of comparison, either with itself or with some external benchmark, such as an environmental quality standard or a guideline value (Brady 2005, Chapter 5 Measurement and Monitoring).

There are a number of ways of estimating the odour release value. Some are likely to have a greater uncertainty associated with them than others (see Table 4.1).

Table 4.1 Different techniques for estimating the odour release value

Estimating odour release from measurement of a specific surrogate odorant compound: e.g. measurement of H ₂ S is commonly used as a surrogate at sewage treatment works; the NH ₃ emissions profile can be used as a surrogate for total odours in certain livestock installations	<p><i>Generally, less uncertainty</i></p>  <p><i>Generally, more uncertainty</i></p>
Using published emissions factors for odour	
Estimating odour release from analyses of bulk materials, e.g. Odour Potential (OP)	
Using calculations and mass balance	
Estimating odour release based solely on professional judgement. (This is a method of last resort.)	

Two important points should be noted about Table 4.1. Firstly, that some *estimates* are obtained using *measurements*! What makes them estimates is that these measurements are not of odour emission directly, but of some other measurand which is then used to infer an odour emission rate. One important aspect of this is the impact of quality assurance/quality control (QA/QC). Generally speaking, the main effect of good QA/QC in minimising uncertainties is where practical work has been used. This really limits its effect to *measurements* of odour emissions directly, or to those odour emissions techniques that use some form of measurement.

Secondly, it is very difficult to give a generalised rank order of these different estimation techniques in order of increasing uncertainty: it depends very much on how suited the chosen technique is to the installation/situation in question. For instance, obtaining an estimate of odour release from measurement of a surrogate compound like hydrogen sulphide (H₂S) or ammonia (NH₃) may be a reliable way to proceed for some processes where a good correlation with odour has been demonstrated and where the published odour detection threshold (ODT) for that compound is reliable, but if this is not the case then the use of published emissions factors for odour (if available) may be better. The ranking shown in Table 4.1 is therefore an approximate one, based on what one may expect intuitively.

There are, therefore, a number of choices that a practitioner needs to make at this phase of the Draft H4 assessment methodology:

- Will the release values be based on measurements or estimations?
- How many samples will be taken for analysis?
- What minimum level of QA/QC will be stipulated for the organisations carrying out the sampling and analysis?

These choices will have a fundamental effect on the uncertainty of the odour release value and will contribute to the overall uncertainty of the exposure prediction. In the remaining sections of Chapter 4 the component uncertainties in the odour release value are considered in greater depth.

4.3 Uncertainties associated with the application of the technique

4.3.1 Temporal uncertainties

There will be temporal uncertainties associated with the application of the technique to the particular situation or scenario, determined by how well the odour release value obtained reflects the timescale of interest. For example, does the measurement or estimate that has been obtained give a good picture of short-term emissions or annual average emissions (as appropriate)?

This component is so situation-specific that it is difficult to put any kind of general figure on its uncertainty. The following factors have a big influence on the magnitude of this component uncertainty and, by considering them, a practitioner can get a qualitative “feel” for their importance.

How emissions vary with time

Odour emissions from single or multiple sources can vary in time in terms of the odour concentration and (sometimes) the character and hedonic tone of the odour, and process parameters can have a major influence on this. Draft H4 recognises that there can be considerable fluctuations even within a single source, depending on a number of factors such as the process cycle, or the

weather. The variability of source strength measurements is discussed in some detail by Best *et al.* (2004).

Large variations in odour emission rates have been found for agricultural sources and intensive farming activities (e.g. poultry units), waste water treatment works and landfill sites. These variations result partly from the biological sources (e.g. age of birds) and the fact that the processes are not operating under constant conditions: for example, the moisture content of poultry litter increases with the age of the birds, leading to a 5-fold increase in the odour emission rate. Additional problems are caused by the extended nature of the source and/or the strong meteorological dependence of decomposition processes: ambient temperature affects the biological processes and barometric pressure can have a significant influence on emission rates. Open-air sources may be affected by rainfall.

For industrial or water treatment sources, changes in feedstock or production cycles often dominate, with some well-defined cyclical characteristics but often with quasi-random components. Odour emissions are also often dependent on operational conditions, which may be difficult to anticipate and control effectively, for example influent load, changes in the nature of the sewage received, frequency of use of open storm tanks, equipment failures, temperature and moisture contents of odorous materials.

Continuous versus periodic monitoring – how well they account for temporal variations

Where the release value has been obtained from measurements (either measurement of odour directly or estimations of odour made from measurement of a surrogate compounds), the type of monitoring – whether continuous or periodic – will be an important factor influencing the size of the temporal uncertainty component. The different approaches for monitoring source emissions are shown in Figure 4.3 and explained in Box 4.2. The various techniques for quantifying odour emissions are summarised in Draft H4 Part 2.

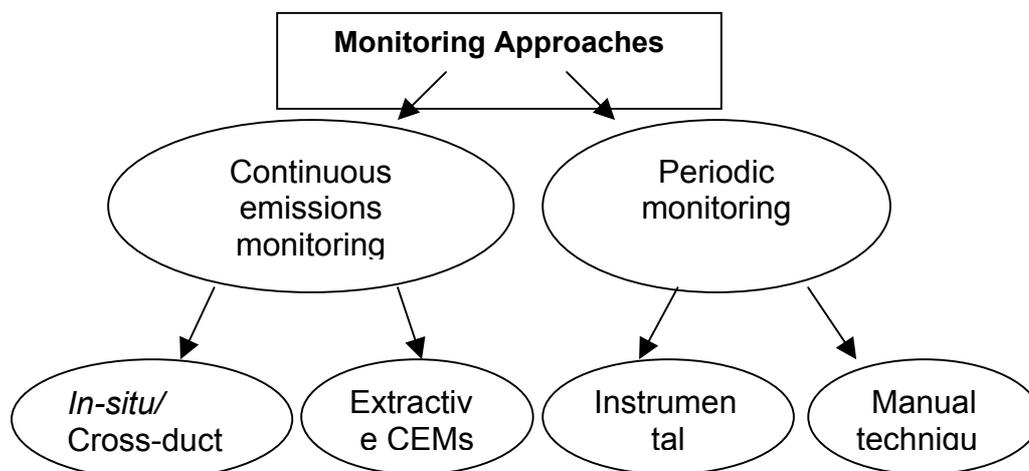


Figure 4.3 Approaches to stack emissions monitoring

Box 4.2 Terminology – periodic and continuous monitoring

Periodic measurements – a measurement campaign is carried out at periodic intervals, e.g. once every three months throughout the year. The sample is usually, but not always, withdrawn from the stack and analysed remotely (extractive sampling). An instrumental/ automatic technique may be used, where the sampling and analysis of the determinand is carried out by a single “black box”. Alternatively, a manual technique may be used where the determinand is sampled on site but is usually analysed later in the laboratory. Samples may be obtained over fairly lengthy periods of several hours, or may be so-called spot samples or grab samples collected over a period of seconds to a few minutes.

Continuous emissions monitoring systems (CEMs) – automatic measurements carried out continuously, with few if any gaps in the data produced. Measurement may be carried out *in situ* in the stack (often called cross-duct monitoring), or extractive sampling may be used

Odour annoyance can be caused by exposures of very short duration – even by episodes lasting a few seconds. If odour exposure is to be modelled in a way that allows correlation with annoyance, it is important to capture the peak emissions in any measurement of source strength. Continuous monitoring using automatic instruments is ideal, as this allows both short-term and annual-average emissions information to be extracted. Unfortunately, continuous emissions monitors (CEMs) do not currently exist for directly measuring odour concentrations (as $\text{ou}_E \text{m}^{-3}$). Hence monitoring of odour concentration directly needs to be carried out using repeated, periodic, manual (i.e. non-automatic) measurements. (This involves a “lung” sampler and analysis by dynamic dilution olfactometry (DDO) to the BS EN 13725 method*, described more fully in Section 4.4.1.2).

For estimates of odour emissions inferred from specific surrogate odorant compounds, such as H_2S or NH_3 , both continuous automatic techniques and periodic manual techniques exist. In situations where there is a good correlation between the surrogate compound and odour, and where the published ODT for that compound is reliable, the reduction in the size of this component uncertainty by having a continuous monitoring may compensate for having to estimate odour instead of measuring it directly.

* BS EN 13725: 2003, Air Quality – Determination of Odour Concentration by Dynamic Olfactometry

Periodic monitoring – designing a sampling strategy to minimise uncertainties from temporal variations

The magnitude of the temporal uncertainties for the application of periodic monitoring to a particular situation will depend on the number and frequency of periodic measurements and their timings in relation to variations in the odour emissions profile of the process. It is not possible to place any numerical estimates of the magnitude of this situation-specific component uncertainty, but practitioners must make themselves aware of its relative magnitude and importance. Annexe J (Informative) *Sampling Strategy* of BS EN 13725 requires that these factors should be taken into account.

For odour assessment purposes, the dispersion model requires emissions data which are *highly-resolved* in time and space because odour annoyance can be caused by short-term peaks in odour concentration. Ideally, the data would be resolved over a few seconds (as this is the timescale of individual odour events) but, as noted above, no CEMs currently exist for directly measuring odour concentrations and we have to rely on periodic measurements. In practice hourly data which take account of diurnal and/or process variations are likely to be the best available. For existing processes, direct source strength measurements at every different stage of an operating regime is the ideal, including any daily, weekly, production or seasonal cycles. However, odour emission measurements are relatively expensive and currently in the UK they are rarely carried out with such frequency. If direct source-strength measurements are only practicable for a limited part of the emissions profile, then this should be the point in the cycle when odour emissions are likely to be highest*. Then, the odour emission rate at other points in the cycle could be estimated (with some care) with values below the measurement maximum.

Draft H4 states that worst-case emission scenarios should be considered in an odour assessment. The majority of controlled, point sources (namely those to which the Draft H4 approach is most applicable) have odour release rates which are not affected by ambient weather conditions. Even if maximum, worst-case emissions can be measured/estimated/predicted, the difficulty for predictive modelling lies in the choice of the prevailing short-term weather conditions for atmospheric dispersion and dilution of the plume that are assumed to coincide with the peak emission episodes.

Project 1 of this Environment Agency cluster (Environment Agency, 2007a) identified well-defined cycles in emissions from many livestock production systems. Bull (2004) investigated the use of different emission scenarios for a chicken rearing facility (a constant emission rate over the entire day and increasing at the end of the 45 day rearing cycle, compared against a scenario allowing diurnal variations). The AERMOD dispersion model was used for this hypothetical installation. It was found that a constant emission rate gave unrealistically pessimistic results for the 98th percentile of hourly mean odour

* This point in the emissions cycle may be identified by repeated, sequential measurements of odour (or some surrogate or indicator), or from a thorough understanding of the process.

concentrations. Allowing diurnal variations gave the lowest results, because it coupled lower emission rates (due to reduced ventilation of the sheds at night) with the worst hourly (stable) meteorological conditions which occur at night.

It is not known whether the historical dose-response modelling studies (for livestock installations), which formed the basis of the Draft H4 Indicative Odour Exposure Standards, took account of emissions variations. No details of the manner in which the odour source data were collated for that study have been found to inform Project 3.

The issue of how well the release value obtained reflects actual emissions over the timescale of interest also applies to release values that are *estimated* using measurements of specific surrogate compounds (e.g. H₂S), emissions factors, analysis of bulk materials (e.g. Odour Potential) and calculations.

4.3.2 Spatial uncertainties

There will be spatial uncertainties associated with the application of the technique to the particular situation or scenario, determined by how well the odour release value obtained at the sampling location reflects what is being released from the stack.

Starting first of all at the large-scale spatial issues, it is not uncommon – especially for plant that has been proposed but not yet built – for the odour release value to be estimated from measurement (of a odour directly or of surrogates) at a representative plant elsewhere. This is combined, if necessary, with factors for scaling up or down. The uncertainty associated with this approach depends on just how representative the plant is, and how valid the scaling factors are. Where odour has been estimated from a surrogate compound (e.g. H₂S), there is also the issue of how strong that correlation is for the site in question. (This is discussed further in 4.4.2.1 for measurement of specific compounds.)

There are also spatial issues with the particular process. For quantifying odour releases from controlled point sources such as stacks, it is preferable to sample the residual odour emission rate after any abatement system such as scrubber, biofilter, etc. However, there may be situations where the odour release value is obtained before an abatement system. The obvious error is introduced by the measured concentration itself not reflecting the concentration being released from the stack, and it would be necessary to correct this figure for the abatement efficiency. This efficiency estimate is itself subject to some uncertainty, which is discussed below. But there is a second possible source of error that can be introduced by using this approach: the hedonic tone of an odour can be substantially altered by some types of abatement. For example, the odour from a rendering process is generally extremely unpleasant; however, if the same odour has passed through a bio-filter it is much less unpleasant (even at the same odour concentration). Odour abatement techniques are described in a number of documents (Environment Agency, 2002c) including Draft H4 Part 2, Stuetz and Frechen (2001), and UK WIR Best Practicable Means Guidebook (2006).

These issues of how well the release value used reflects actual emissions being released from the stack also apply – to a greater or lesser extent, depending on the situation - to release values that are *estimated* using measurements of specific surrogate compounds (e.g. H₂S), other surrogate measurands, emissions factors, and calculations. For example, when using emissions factors these will usually give unit emission rates for the unabated source. This value will need to be corrected for the efficiency of any abatement that is used. This efficiency value will probably be based on manufacturer's figures or published, generic abatement efficiencies – if the efficiency was measured by odour sampling then there would not be a need to use emissions factors*. However, such published performance information may not have been based on measurements of total odour before and after the abatement, and may be based on some specific odour compounds only (e.g. H₂S). Such efficiency values may not make allowance for the other odorous components in the total odour of the site in question. Furthermore, the performance characteristics of certain abatement techniques depend on ambient temperature. There is, therefore, a need for the practitioner to question the reliability of the published abatement efficiencies.

Again, it is not possible to place any numerical estimates of the magnitude of these situation-specific uncertainties, but practitioners must make themselves aware of their possible importance, and form a qualitative view on the significance of this component of uncertainty.

4.4 Uncertainties associated with the technique itself

4.4.1 Component uncertainties when measurements of odour emissions are used

4.4.1.1 *General overview of uncertainties in source emissions measurements*

There are a number of useful sources of information on uncertainty. General guidance on uncertainty has been published by the International Standards Organisation (ISO 1995), while other guidance from Eurochem (1995) and the Royal Society of Chemistry (Farrant 1997) addresses uncertainty as applied to analytical measurements. Several years ago, the Source Testing Association (STA) produced guidance for stack emissions monitoring specifically (Pullen

* Where the abatement efficiency value has been obtained from measurement, the quality of that measurement is important. Ideally odour should have been measured by DDO to method BS EN 13725. The repeatability of this measurement is an important consideration: a large number of replicates improves the certainty in the measurement. This is discussed in more detail in Section 4.4.1.2.

1998). The underlying principles of this guidance remain valid today, and the STA has reviewed how these main principles should be applied to new and emerging standards and developments and changes in the existing published standards that are used to carry out stack monitoring (Pullen 2003).

The main principles of estimating uncertainty in stack monitoring are described in detail in Appendix A, including calculating uncertainties in practice and combining component uncertainties to give an estimate of the overall uncertainty. This section focuses on some key points relevant to the identification of component uncertainties for quantification of odour specifically. Both measurement of the odour emission itself ($ou_E s^{-1}$), and estimation of odour emission from measurement of a specific, surrogate compound (e.g. $mgH_2S s^{-1}$) are described here.

Though there are a number of different methods of assessing uncertainty. The general principle is to use the following process:

- i. **Specification** - define what is being measured and the parameters on which it depends;
- ii. **Identify the sources of uncertainty** - for each parameter in the relationship, list the possible sources of uncertainty, e.g. sampling, instrument bias, reagent purity, environmental conditions;
- iii. **Quantify the uncertainty** - measure or estimate, then calculate total uncertainty using accepted rules.

i. Specification of the measurement

Quantification of the odour release from a controlled source involves measuring the odour concentration and the volume flow of the effluent gas (Equation 4.1):

Equation 4.1

<p>Odour release rate</p> <p>$ou_E s^{-1}$</p> <p>or</p> <p>$mg [compound] s^{-1}$</p>	=	<p>Odour concentration</p> <p>$ou_E m^{-3}$</p> <p>or</p> <p>$mg [compound] m^{-3}$</p>	x	<p>Volume flow</p> <p>$Nm^3 s^{-1}$</p>
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ii Specification and identification of the sources of uncertainty

The overall uncertainty that should be considered is that of the whole measurement - made up of a sampling stage and an analytical stage*. The sampling and analysis equipment will be an important, but not the only, component of this. In this conceptual model, measurements using direct-reading instruments (e.g. for H_2S) should be treated no differently to measurements where a sample is collected discretely and then analysed later

* It is important to not use restrictive definitions of sampling and analysis: in this context they mean the collection of the measured property and its quantification, respectively. Sampling can include, for example, the placing of a thermocouple and pitot at a particular point in a stack to obtain the information to calculate the volume flow (necessary to get the mass odour emission rate from the concentration). The analysis stage in this example is the instrumental conversion of the physical properties to electronic readouts of temperature and pressure.

(e.g. by DDO) on the laboratory bench. Both have sampling and analysis stages: for example, a flame photometric detector (FPD) analyser for continuous monitoring of H₂S samples the stack gas through a heated line, conditions the gas (removal of solids and sometimes moisture) and then presents it to the reaction cell where the analysis takes place^{**}.

Both the sampling and the analysis have themselves many component uncertainties that go to make up the overall uncertainty. Some examples are shown in Figure 4.4. This list is not exhaustive, and careful thought needs to be given to the potential sources of uncertainty in the method at each of these stages.

This conceptual model is not specific to one class of technique: it is applicable to odour measurements made using periodic monitoring (e.g. lung sampling followed by DDO), using continuous emissions monitors (CEMs) (e.g. an FPD analyser for measurement of H₂S), or using classical techniques. Here again, it is helpful to avoid interpreting terms too restrictively. A number of examples follow:

- Losses during transport and conditioning of the sample would apply equally to losses due to degradation and adsorption on container walls during transport of a bag sample solution from a periodic stack test to a laboratory for later DDO analysis^{***}, and to errors from condensation and adsorption in the heated line of a continuous analyser;
- Sampling collection medium efficiency would apply equally to the collection efficiency of a solid-sorbent resin such as Tenax used to measure specific surrogate compounds of the odour (e.g. H₂S), and to the collection efficiency of a sorbent solution in an impinger if a classical sampling technique were used;
- For analysis, calibration uncertainties for DDO would include the uncertainty in the standard reference odour (see Table 4.1) due to the tolerance of the reagent purity. For an instrumental analyser used to measure a specific surrogate compound the calibration error could include the tolerance on the traceable calibration gas standard used;

^{**} Light emitted by the excited molecules is filtered through a narrow bandpass filter to allow detection at the appropriate wavelength (394 nm for sulphur) using a photomultiplier tube. Calibration is by standard concentrations of compressed SO₂ gas and pure air.

^{***} Measurement of an odour with a time lag between sampling and analysis is termed *delayed olfactometry* (BS EN 13725: 2003) and is much more common for source emission quantification than *direct olfactometry* (also known as *on-line olfactometry*), where there is no time lag.

- Volume errors for DDO analysis would include the tolerance in the dilution apparatus. For instrumental techniques to measure surrogate compounds it would include tolerances on reaction-cell volumes, sample-loop volumes, etc. For a classical wet analysis it would include the tolerances on the volumetric glassware used (e.g. measuring cylinders, pipettes, burettes, etc.).

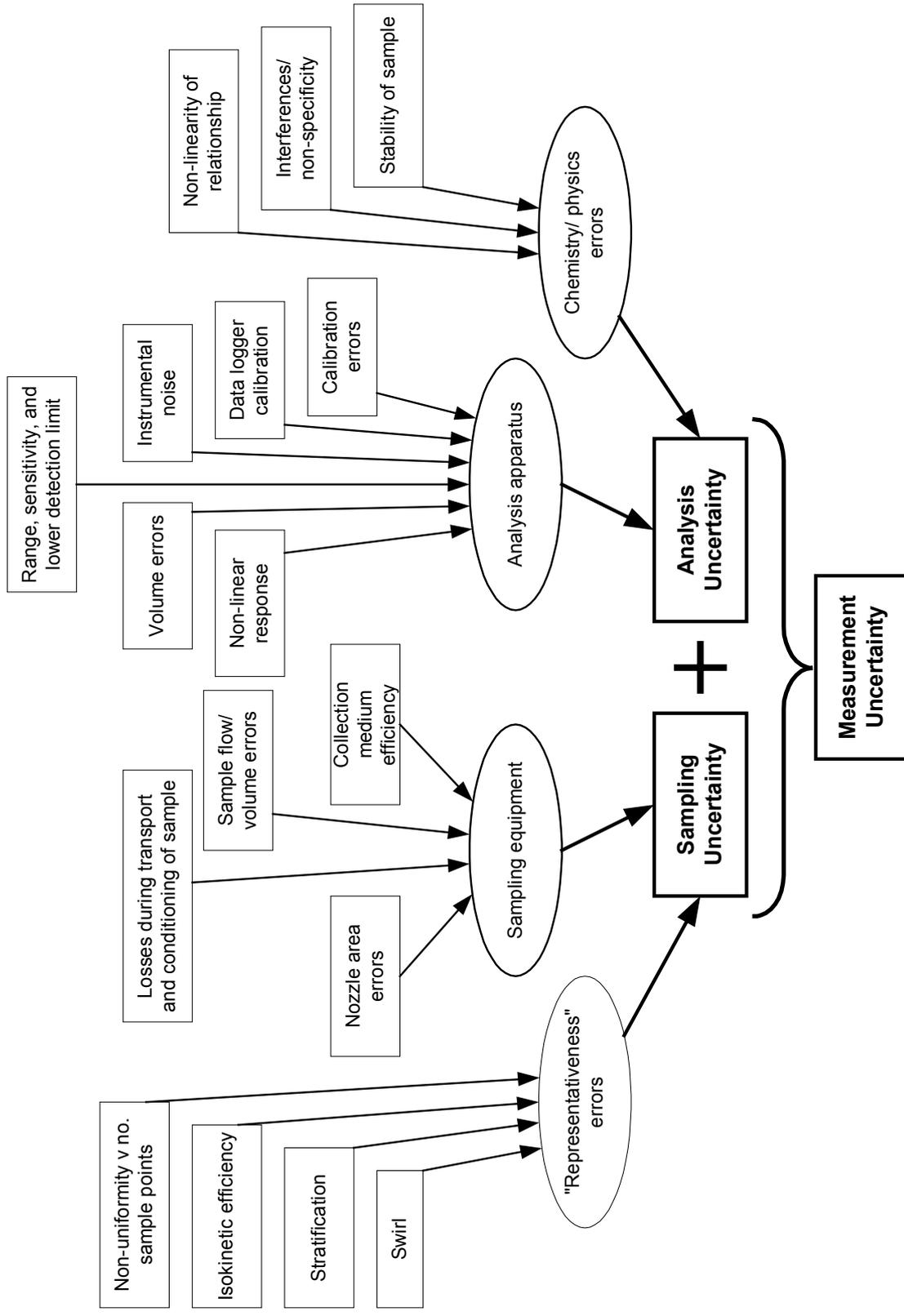


Figure 4.4. Examples of component uncertainties contributing to the overall measurement uncertainty

iii Quantifying the uncertainties

There are four main approaches to quantifying the uncertainties, summarised in Table 4.2. Different approaches are appropriate to different situations.

Furthermore, some of these approaches give better estimates of uncertainty than others: the best provide information on the combined effects of the reproducibility of the measurements and the bias (the difference between a measured value and the “true” value); others give a reasonable estimate of the uncertainty arising from random effects (scatter) but do not address systematic, or bias, uncertainties.

Table 4.2 The four main approaches to quantifying uncertainties in source emissions measurements

Approach	Advantages	Disadvantages
A. Repeat measurements on a certified reference material (CRM)	Gives a “real” value for the whole uncertainty (i.e. the combination of precision and bias)	The accepted reference value for the European odour unit (ou _E) is the European Reference Odour Mass (EROM), which is equal to a 123 µg <i>n</i> -butanol evaporated in 1 m ³ neutral gas, which produces a concentration of 0.040 µmol/mol. Repeat DDO measurements have been carried out using this standard in the laboratory, but this only covers the analysis aspect of the measurement and not the sampling. A standard reference odour (or surrogate, e.g. H ₂ S) in the gas stream of a stack is not available, so this approach cannot be used currently to quantify the sampling component of the measurement uncertainty.
B. Experimental work, e.g. repeatability experiments, paired comparisons, and ring tests	Gives a “real” value for the precision component of the uncertainty	Gives good estimate of precision, repeatability, etc. but often fails to include bias. This approach can be expensive, e.g. CEN ring tests up to €100,000 per test with multiple teams
C. Uncertainty Budget Approach - estimations based on previous results/data, e.g. instrument specifications, calibration data, Proficiency Testing schemes	Can “unpick” the uncertainty budget and rebuild if there are changes, e.g. deviations, new equipment.	Can appear rather abstract.
D. Estimations based on considered judgement	This is usually considered to be the method of last resort	

Note: for definitions of terms such as precision, repeatability, reproducibility, accuracy, bias, and uncertainty refer to the definitions given in the CEN standard EN 13725.

As explained in Chapter 1, the purpose of this study is not so much to get a precise estimate of the overall uncertainty, but rather to investigate the relative importance of the main component uncertainties. It would be useful to know, for instance, if dispersion modelling has a much more significant component uncertainty than emissions quantification, or assignment to different bands of unpleasantness. To do this we need to examine separately – quantifying if possible – each of the component uncertainties. We then need to see how they compare. For this, the Uncertainty Budget Approach (C) is relevant.

4.4.1.2 *Uncertainties for direct measurement of odour emission concentration ($ou_E m^{-3}$)*

The main principles of the measurement

The procedure for determining the odour concentration by DDO in a source emission is specified in BS EN 12725. The main principles of the measurement are as follows:

A gaseous sample of the odorants is generally* obtained using the “lung principle”, where the inert sample bag is contained in a rigid container (e.g. a barrel) that is evacuated by a pump. The drop in pressure in the space between the sample bag and the container causes the sample bag to fill from the source.

For sampling of controlled point sources, an inert sample line goes from the sample bag to a probe inserted in the stack or duct. Pre-dilution is used where there is a risk of condensation of the sample when stored at ambient conditions and where the sample gas is very hot and needs to be cooled before entering the sample container. Two techniques can be used for this: (i) static pre-dilution, where the sample bag is filled with a known volume of dry, odour-free air or nitrogen before the sample gas is collected, and (ii) dynamic pre-dilution, which uses a flow-control device to mix sample gas with neutral gas during sampling.

The analytical stage consists of presenting, in the laboratory, a panel of selected and screened human subjects with that sample. The human subjects (or more specifically the response of the olfactory organs of the subjects) are essentially the sensors of this analytical technique. The concentration is varied by diluting the sample with neutral gas in order to determine the dilution factor at the 50% detection threshold. At that dilution factor the odour concentration is, by definition, one European odour unit ($1 ou_E m^{-3}$). The odour concentration of the original, undiluted sample is then expressed as a multiple (equal to the dilution factor at the 50% detection threshold) of one European odour unit per cubic metre at standard conditions for olfactometry. For example if one hundred dilutions were needed to reach the 50% detection threshold, the odour concentration would be $100 ou_E m^{-3}$.

The basis of traceability of this analysis is the linkage with the European Reference Odour Mass (EROM). This, the accepted reference value for $1 ou_E m^{-3}$, is equal to $123 \mu g$ *n*-butanol evaporated in $1 m^3$ neutral gas, which produces a concentration of $0.040 \mu mol/mol$. It means that measured odour concentrations are effectively expressed in terms of “*n*-butanol mass equivalents”. The assumption is made that the precision for olfactometric determination of the reference material, *n*-butanol, is transferable to determinations on non-reference material samples, i.e. source odour samples. The lower limit of detection for a sample using DDO is $50 ou_E m^{-3}$.

* The standard allows the alternative sampling technique of direct pumping into a sample bag, but this technique needs to be used with caution because of the risk of contamination.

The major sources of error in this type of measurement are reported to be the potential for degradation of the sample after it is collected and the variation in performance of the odour panellists (Minnesota Pollution Control Agency 2006).

The major component sources of uncertainty in the DDO analysis

The standard contains performance quality requirements for:

- selecting odour panel members with appropriate individual variability and sensitivity;
- calibration of the dilution equipment (the dilution olfactometer) using a tracer gas and a physical/chemical method of analysis;
- calibration of the sensor (odour panel) by means of the reference odorant.

The standard also specifies:

- requirements for construction of the olfactometer and the materials that can be used;
- conditioning and cleaning of apparatus;
- the dilution gas and reference material to be used;
- the environmental conditions under which the olfactometry is carried out.

The laboratory should also carry out an overall check on precision by means of regular performance testing – an inter-laboratory proficiency testing scheme.

The quoted uncertainty (see below) can therefore be considered to include the component uncertainties relating to the above when the olfactory testing laboratory complies with all the quality criteria specified in the standard.

BS EN 13725 sets several quality performance criteria that the olfactometry laboratory must comply with, including achieving a certain level of repeatability for the determination of the standard odour. Annexe G (Informative) of the standard shows how the standard deviation of a population of test results can be calculated. When the DDO analysis is carried out in full compliance with the standard, using a properly selected panel and achieving the repeatability criterion in the standard, the 95% confidence interval for one single odour concentration determination on a sample of $1000 \text{ ou}_E \text{ m}^{-3}$ will be 453 to $2209 \text{ ou}_E \text{ m}^{-3}$. Analysing replicate samples can reduce the uncertainty and Table G.1 in Annexe G of the standard shows how this affects the confidence interval. A simplified version of this table is given here (Table 4.3). For a concentration value of $1000 \text{ ou}_E \text{ m}^{-3}$ obtained from two DDO determinations, in 95% of cases the “true” result will lie in the interval between $571 \text{ ou}_E \text{ m}^{-3}$ and $1752 \text{ ou}_E \text{ m}^{-3}$. The earlier Environment Agency research (2002b) that formed the basis for the Draft H4 approach summarises graphically (shown here as Figure 4.5) the analysis component of uncertainty that can be expected for a result (nominally $1000 \text{ ou}_E \text{ m}^{-3}$) obtained from a given number of replicate DDO determinations.

Figure 4.5 illustrates well the dramatic improvement in analysis uncertainty that results from basing the result on triplicate determinations. For an analysis result of 1000 based on triplicate determinations the expanded uncertainty (i.e. 95% confidence) expressed as a percentage is -37% on the lower limit and + 58% on the upper limit.

Table 4.3 The 95% confidence intervals for mean DDO results obtained from different numbers of replicate determinations

DDO result (ou _E m ⁻³)	Number of samples (n) analysed for result	Lower limit (ou _E m ⁻³)	Upper limit (ou _E m ⁻³)
1000	1	453	2209
1000	2	571	1752
1000	3	633	1580
1000	4	673	1486
1000	5	702	1425
1000	6	724	1382
1000	7	741	1349
1000	8	756	1323
1000	9	768	1302
1000	10	778	1285

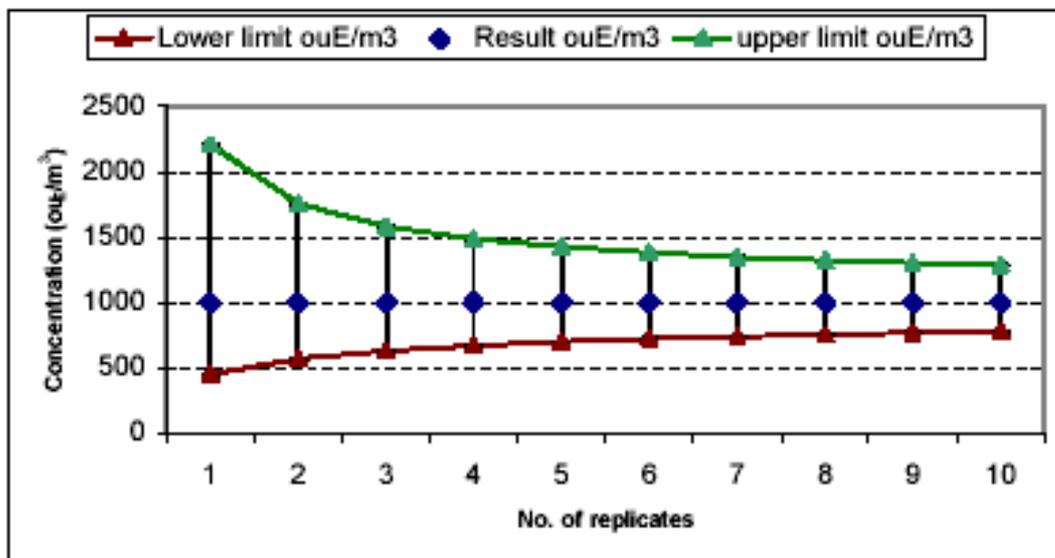


Figure 4.5 The 95% confidence intervals for a mean DDO result of 1000 ou_E m⁻³ obtained from different numbers of replicate determinations

The major component sources of uncertainty in the sampling of the odour

The previously described component uncertainty for DDO quoted in the standard are for only the analysis part of the measurement; the uncertainty value doesn't include the contribution from collecting the stack (or other source) sample. This may be an important contributor to the overall measurement uncertainty.

A major part of the sampling uncertainty will depend on the representivity of the sample collected from the sampling point duct in relation to the bulk effluent gas. If at the sampling location, the bulk gas is well mixed, then these errors will be minimised. If there is stratified gas flow, then the error can become large unless multi-point sampling across the duct is employed. The magnitude of this component uncertainty will be very site-specific and no definitive value can be put on it. However, its significance will be minimised if the sampling location and the number of sampling points conform to the requirements of the Environment Agency's M1 guidance (2006) for regulatory compliance monitoring.

A standard reference odour in the gas stream of a stack is not available, so it is currently not possible to use the type of approach where the sampling component of the measurement uncertainty is quantified. The standard notes that further research is needed to characterise the sampling uncertainty, which was limited by financial constraints when the standard was developed. Improvements in sampling may be the subject of a future revision of the standard. It should be noted, however, that BS EN 13725 contains an acceptance limit for "accuracy" of ≤ 0.20 for dilution apparatus - both the apparatus used in the olfactometer itself and also when pre-dilution is used in sampling to prevent condensation. In the absence of any other estimate for the sampling component uncertainty, this figure of ± 0.20 or $\pm 20\%$ has been used as a typical value for sampling.

As well as the collection of the sample, transport and storage between sampling and olfactometry can potentially cause deterioration and losses to the sample by adsorption, diffusion and chemical transformation. The standard does cover some of these aspects of sampling: performance quality requirements are set for the materials that can be used in sampling and for conditioning and cleaning of apparatus. The standard notes that experiments so far indicate that losses become significant after 24-30 hours of storage for some substances and therefore the standard requires that the interval between sampling and analysis shall not exceed 30 hours and the sample shall be kept at $>\text{dewpoint} < 25^{\circ}\text{C}$ and out of direct sunlight. The assumption is made here that the additional sampling uncertainties due to transport and storage are not significant if these requirements of the Standard are complied with.

* This was stated as taking into account both bias and random errors, and expressed as repeatability at 95% confidence limits

4.4.1.3 Uncertainties for measurement of volume flow ($\text{Nm}^3 \text{s}^{-1}$)

In order to calculate the odour mass emission rate from the odour concentration measurement, it is necessary to know the volume flow.

Annex I (Informative) of BS EN 13725 quotes method ISO 9096* for measuring the flow rate in the duct, with the volume flow rate then calculated at 'Standard' conditions (293 K, 101.3 kPa) for wet conditions following ISO 10780**.

However, atmospheric dispersion models require the volume flow at Normal conditions (273 K, 101.3 kPa) rather than Standard conditions. This is obtained from the following equation:

Equation 4.2

Volume flow	=	Gas stream velocity	x	Duct cross sectional area	x	Temperature correction	x	Pressure correction
$\text{Nm}^3 \text{s}^{-1}$		m s^{-1}		m^2		$273/\text{Temp(K)}$		Pressure (kPa)/101.3

The actual measurands to obtain these factors are:

- Pitot pressure (kPa), calibrated against velocity;
- Duct diameter, or breadth and depth (m); and
- Thermocouple output (mV) calibrated against temperature (K).
- Pitot pressure (Pa), giving absolute pressure in the stack.

Each of these has their own component uncertainties.

Hawksley *et al* (1997) studied measurements of particulates to BS3405. Although this standard is now withdrawn, the flow measurement aspect is little different from the more recent standards. Using the flow estimate to calculate the particulate concentration from the particulate emission rate was said to increase the uncertainty by a $\sqrt{2}$ factor.

As part of its MCERTS accreditation, Bureau Veritas' emissions monitoring team has calculated the uncertainty of gas flow rate measurements using pitot static tubes for a typical stack of duct diameter 1 m and gas velocity of 15 m s^{-1} (Bureau Veritas 2005). Based on an uncertainty budget built up from the component uncertainties of velocity measurement and linear dimension measurement, the standard uncertainty is about $\pm 5.7\%$ of the calculated value, and the expanded uncertainty at 95% confidence limits (i.e. x 1.96) is about $\pm 11\%$. This does not include uncertainties introduced by conversion of the volumetric flow-rate at duct conditions (of temperature and pressure) to

* ISO 9096, Stationary source emissions – determination of concentration and mass flow rate of particulate material in gas-carrying ducts – Manual gravimetric method.

** ISO 10780, Stationary source emissions – Measurement of velocity and volume flow-rate of gas streams in ducts.

reference conditions. Bureau Veritas calculates that the temperature measurement typically has an expanded uncertainty of $\pm 5\%$. Taking the square root of the sum of the squares of these component uncertainties gives a combined standard uncertainty of $\pm 6.2\%$, or an expanded uncertainty of $\pm 12\%$. This assumes the sampling location and stack gas flow stability meets the requirements of the Environment Agency's M1 guidance for regulatory compliance monitoring. This also enables the uncertainty due to the measurement of volume flow (which is an input to the dispersion model) to be ranked relative to other component uncertainties for Phase II (see Figure 5.2).

The component uncertainty for flow measurement also applies to estimates of odour releases that have been obtained from measurements of specific, surrogate compounds (Section 4.4.2.1). Emissions factors and mass flow calculations (Sections 4.4.2.2 and 4.4.2.4, respectively) usually require knowledge of flow as well.

4.4.1.4 *Minimisation of measurement uncertainties by effective QA/QC*

High standards of QA/QC are essential in measurements of odour emission rates. When an emission rate is used in dispersion modelling, any error in the emission rate will transfer directly to an error in the predicted 98th percentile ground level odour concentration. Sampling and analysis should be in accordance with accredited procedures, for which error margins have been assigned.

At a general level, all emissions measurements submitted to the Environment Agency for regulatory compliance purposes must be covered by the Environment Agency-sponsored MCERTS quality scheme (see Box 4.3).

Box 4.3 MCERTS

The Agency's Monitoring Certification Scheme (MCERTS), launched in 1998, focused initially on type approval of continuous monitoring systems (CEMs) for stacks. The Agency has broadened the scheme to cover periodic measurements using portable equipment, termed manual stack-emissions monitoring. Because the quality of such periodic measurements depend not only on the equipment, but also on the competence of the individual and the organisation, MCERTS for manual stack monitoring covers both these areas.

Organisations and individuals carrying out the measurements must hold MCERTS certification and qualifications, respectively. The organisation must meet the requirements of EN ISO/IEC 17025 for the MCERTS Performance Standard for Organisations. This provides a robust framework for the quality of the measurements and helps ensure uncertainties are minimised. At a method-specific level, BS EN 13725 contains numerous QA/QC requirements to ensure that the testing laboratory produces olfactometry results within the quoted uncertainty for the DDO method.

The use of QA/QC to minimise uncertainties applies not only to DDO measurements but also to estimates of odour releases that have been obtained from measurements of specific, surrogate compounds (Section 4.4.2.1). Again,

MCERTS should apply where the measurements are for regulatory compliance purposes.

4.4.1.5 *Comment on odour measurement uncertainties for diffuse and fugitive sources*

Diffuse sources

Although emission rates of area sources (both solid and liquid) can be measured, ELVs have not been widely used for compliance checking purposes (although measurement is sometimes used for impact assessment). As such releases are not usually controlled sources, they generally cannot be operated in such a way as to remain below an imposed ELV; instead these sources tend to be managed by indicative/generic Best Available Techniques (BAT) standards to minimise releases. As the Draft H4 approach is used mainly in PPC applications to back-calculate an acceptable ELV for day-to-day source control from the ambient odour benchmark, it finds little application in regulatory compliance of diffuse sources.

Nevertheless, for completeness, some comments on the measurement uncertainties for diffuse sources are given here. It is possible to measure the emission from area sources such as open tanks, but only if the surface is homogeneous. The analytical stage, involving DDO, is common for any source odour sample; however, significant differences exist between the techniques for collecting samples from diffuse sources compared to point sources. Gostelow (2000) describes the special techniques (e.g. micrometeorological method, flux hoods) to estimate odour emission rates from area sources but points out that further work is required to validate many odour formation prediction calculations. BS EN 13725 describes how area sources are sampled: instead of using an inert sample line to a probe in a stack, the sampling line goes to a sampling hood which is placed over, or floated on, a defined surface area of the source (e.g. landfill surface, effluent lagoon, etc.). The hood is ventilated with odour-free air at a known volumetric flow rate. The velocity of this ventilation air across the sampled area of the source affects the rate of odour emission from the source, and must therefore be noted. This arrangement is known as a “flux hood” and several types exist with considerable variation in design. One common type is the Lindvall hood, but the standard points out that this was originally designed only for making comparative measurements of transfer rates from surfaces; there is a need for standardisation and a re-assessment of the underlying physical assumptions, including whether the flow pattern simulated under the hood is representative of real conditions (as is commonly assumed). In particular, the air velocity used is often well below typical wind velocities. There is also the potential that interaction with the matrix (e.g. shading of the source/protection from the wind etc.) could alter evaporation. The standard states that further research and validation is required.

No details have been found on how these limitations translate into numerical sampling uncertainties. However, it is reasonable to assume that sampling of odour from a diffuse release such as an area source has a greater level of uncertainty than for a controlled point source (e.g. stack) release. The reasons for this are:

- Sampling of odour emissions are far easier for stacks than for area sources;
- Sampling of odour emissions from stacks is more representative of real conditions than sampling from area sources;
- Area sources are inherently more prone to emissions variability e.g. seasonal variability for lagoons at waste water treatment facilities.

Fugitive sources

Fugitive emissions can occur from plant, buildings, tanks and vessels. Fugitive odour events are often a result of infrequent operations (such as cleaning or maintenance) or continuous releases (such as leaks from valves and pumps at a refinery) which produce high levels of odour yet cannot be quantified or measured. Examples of fugitive emissions or those which cannot be readily measured are:

- Raw material or waste storage and handling;
- Transfer processes (e.g. unloading of vehicles at waste transfer site);
- Washing, cleaning and maintenance operations;
- Start-up and shut-down;
- Emergency releases or leaks;
- Pumps and flanges;
- Formation of odorous by-products;
- Contaminated or blocked drains.

BS EN 13725 states clearly in Annexe J (Informative) *Sampling Strategy*, that quantifying fugitive emissions incurs large errors and no known sampling technique can be recommended. No figures are given, but it is reasonable to deduce from this that they are in orders of magnitude. Draft H4 recognises that fugitive emissions are difficult to measure/predict, and their modelling in most cases is inappropriate. Fugitive emissions are better controlled through improved housekeeping/containment and by indicative/generic BAT, than by application of the modelling/back-calculation approach.

Other acute odour emissions that can be considered objectionable or offensive on a single occasion are often associated with abnormal or upset conditions. Examples are malfunctioning abatement systems, or infrequent activities such as re-opening old areas of fill at landfill. Such highly variable and/or uncontrolled discharges are typically very difficult to quantify and as such are not amenable to the Draft H4 predictive approach.

Again, there are no details on how these limitations translate into numerical sampling uncertainties. However, bearing in mind the above comments and recommendations it is reasonable to assume that sampling of odour from a fugitive release would have a significantly greater level of uncertainty than for either a controlled point source (e.g. stack) release or a diffuse release such as an area source.

4.4.2 Component uncertainties when estimates of odour emissions are used

4.4.2.1 *Uncertainties from odour release estimates derived from measurement of specific compound concentrations (mg m^{-3})*

Application of this approach

It is possible to estimate the odour release value from measurement of a surrogate compound that is correlated well with odour. The use of odour detection threshold (ODT) values for specific chemical species to effect a conversion from the concentration of the compound (mg m^{-3}) to an equivalent odour concentration ($\text{ou}_E \text{m}^{-3}$) is described in Draft H4. The equivalent odour concentration is then multiplied by the volume flow of the release to give the equivalent odour emission rate (ou s^{-1}).

This approach has been used at sewage treatment works and for intensive poultry farming emissions, where popular surrogates for odour have been H_2S and NH_3 , respectively.

There are three aspects to the uncertainty in estimating odour emissions from specific compounds:

- What is the uncertainty in the measurement (sampling plus analysis) of the surrogate compound?
- How well does the specific compound account for the total odour at the site in question? This was discussed briefly in Section 4.3.2.
- How robust is the conversion from the concentration of the compound (mg m^{-3}) to an equivalent odour concentration ($\text{ou}_E \text{m}^{-3}$) using the ODT?

Uncertainty in the measurement of the surrogate compound

The Environment Agency's M2 guidance (2004) lists the monitoring techniques and published methods that are acceptable under MCERTS for regulatory compliance checking of controlled point-source emissions to air. For H_2S and NH_3 , both continuous automatic techniques and periodic manual techniques exist, operating on a variety of analytical principles such as uv and ir-spectrometry, chemiluminescence, continuously-sampling chromatography, FPD, and electrochemical cells. For some diffuse sources, continuous automatic techniques can also be used: in Project 1 of this cluster (Environment Agency 2007a), Silsoe Research Institute describes techniques it has developed to monitor continuously the emissions ammonia from agricultural buildings. In situations where there is a good correlation between the surrogate compound and odour, the reduction in the size of this component uncertainty by having a continuous monitoring may compensate for having to estimate odour instead of measuring it directly.

The measurement uncertainty for each of these techniques and methods will be somewhat different (and this information is not listed in M2). All MCERTS-accredited organisations carrying out such measurements for regulatory compliance purposes are obliged to quote the measurement uncertainties, incorporating sampling and analysis. For modern instrumental techniques the

expanded (i.e. 95% confidence level) uncertainty of concentration measurements is usually within $\pm 10\%$ (expressed as a percentage of the measured range).

This is consistent with values quoted by Yang and Hobson (2000) who reviewed techniques for estimating emission rates of H_2S from point sources, from area sources (channels, weirs and open tanks) at waste water treatment works using a flux chamber, and from estimations based on the theory of gas-liquid transfer. The error associated with the determination of odour emission rate for a point source was said to be typically around $\pm 3\text{--}10\%$ for H_2S if the measurement of airflow is accurate. The authors recognised that little is known about the accuracy of indirect methods for estimating odour emission rates. This study also stated that odour emission rates from enclosed processes could vary by over two orders of magnitude depending on the condition of the flow.

How well the specific compound accounts for total odour at the site

The specific compound(s) should be responsible for the vast majority of the odour in the emissions, or at least be considered as an empirical surrogate for the odour.

Although there is no absolute consensus on how odour should be quantified at wastewater treatment works, hydrogen sulphide is commonly used as a surrogate for odour due to its ease of monitoring. This technique is acknowledged in the draft Defra guidance (2004) on odour from sewage treatment works. The main limitation of this approach is that one chemical compound cannot always accurately and consistently represent the overall odour effect of a mixture of compounds. For example, odour monitoring studies at some sewage treatment works have shown H_2S to be a poorer indicator of sewage treatment odours than is commonly assumed. Whilst it is true that H_2S may be a major component of the sewage treatment works odour cocktail, this does not allow for the complex organic odours associated with sewage and there are other odorous compounds which may be present and may contribute to the odour impact. Sometimes, a high correlation is found between ambient odour ($\text{ou}_E \text{ m}^{-3}$) and H_2S mass emission rate. On the other hand, one study (Fraser 2004) showed that measured H_2S in liquor samples from preliminary tanks at a waste water treatment works was less than 5% of total odour, which would suggest that under these circumstances H_2S may not be used as a proxy at low levels of odour. Other studies (Stuetz & Frechen, 2001) showed that H_2S was a good surrogate indicator for some aspects of the sewage treatment process, namely sludge storage and handling, and preliminary treatment before odour control, where it is the dominant odorant. Poorer correlation was found when lime dosing is employed, after odour control (which may remove H_2S preferentially to other odorants), for secondary treatment odours/aeration tank odours (which should not generate H_2S unless overloaded), for industrial effluents, and for dryers/incinerators. H_2S is proportionally less important as an odorous component in these sources. It is theoretically possible to use these correlations predict odour concentrations from H_2S concentrations but this would require detailed knowledge of the processes and conditions at the works.

Similarly, other studies (Freeman *et al.* 2000) have found that hydrogen sulphide and methane (CH_4) concentrations in samples of landfill gas did not

correlate well with odour concentration, suggesting that other compounds in the landfill gas were also contributing to the odour nuisance.

It is noteworthy that the Western Australian EPA (Department of Environment Protection 2002) allows the geometric mean air odour threshold to be used for modelling odour impacts only when a single odorant in an air stream is present and there are appropriately reviewed odour thresholds for the odorant available, giving as an example those from the US EPA (1992).

It is not possible to place any generally applicable numerical estimates of the component uncertainty from how well the specific compound accounts for the total odour at the site in question. This component uncertainty is site and situation-specific. Practitioners must make themselves aware of the possible importance and form a qualitative view on the significance of this component of uncertainty.

The robustness of the conversion from the compound concentration to an equivalent odour concentration using the ODT

Even if the surrogate compound is responsible for the vast majority of the odour in the emissions, monitoring of individual compounds is still an imperfect way of quantifying odour releases due to the following:

1. This approach assumes the relationship between gas concentration and odour concentration to be linear, which is not always the case.
2. An odorous gas can comprise a cocktail of many odorous compounds and this approach does not work well for mixtures. Firstly, it is difficult to identify all the odorous compounds. Secondly, the overall odour concentration of a mixture cannot be estimated by simply adding the values of the chemical constituents. This may give an overestimate or an underestimate because there may be non-linear additive or synergistic effects between the various compounds and due to the way that odour stimuli are processed by the human brain.
3. Published odour threshold data may be contradictory and of varying quality. This is due to differences in olfactometry analytical techniques used by laboratories in the past (although this became more standardised from the late 1990s), and also to the use of different definitions of the odour threshold, such as detection, certainty and recognition levels. More details on this are given in the Literature Review for Project 2, which is part of this cluster (Environment Agency 2007b).

The component uncertainty from the conversion from the mg m^{-3} to an equivalent $\text{ou}_E \text{m}^{-3}$ using the ODT depends on the quality of data available for the particular compound. Older data may be less precise and may also contain significant bias, so it is important that practitioners use, wherever possible, up-to-date threshold data obtained using a validated standard method (especially BS EN 13725) to keep this uncertainty to a minimum. Where ODTs have been obtained using DDO carried out to BS EN 13725 then the analysis uncertainty quoted in Section 4.4.1.2 should apply. For example, for an ODT result obtained from triplicate determinations the expanded uncertainty (i.e. 95% confidence) expressed as a percentage is -37% on the lower limit and + 58% on the upper limit.

Other ODTs may have been obtained by methods having less-robust quality criteria, which means different estimates of the ODT values may be reported by different workers. Woodfield & Hall (1994) categorised the reported thresholds for numerous compounds in terms of their likely measurement quality. Even for those DDO analyses with the top three quality rankings – that had been made using a recognised DDO technique* - the range in reported values could span several orders of magnitude. For example, Table 4.4 shows the first five compounds (quoted in alphabetical order) with the ranges of their reported ODT values.

Table 4.4. Some compounds, with the ranges of their reported ODT values

Chemical	Reported threshold range (mg m ⁻³)	Best estimate threshold value (mg m ⁻³)
Acetic acid	0.025 - 0.064	0.043
Acetone	1.4 - 40.2	13.9
Amyl acetate	0.27 – 0.96	0.95
Benzene	1.5 – 108	32.5
1,3- Butadiene	0.45 – 1.1	1.1

For these older data, it is not possible to place any generally-applicable numerical estimates of the component uncertainty from the conversion from the mg m⁻³ to an equivalent ou_E m⁻³ using the ODT. However, it is clear from the handful of values shown in Table 4.4. that it can be large and can have a significant influence on the overall uncertainty. This is an important point, and it is essential that the practitioner considers the quality of the ODT values and forms a view on its significance for the application in question. This may require investigating the provenance of the data, in particular whether the DDO determinations met the requirements of BS EN 13725.

4.4.2.2 *Uncertainties from odour release estimates derived from emissions factors*

Application of this approach

The use of emissions factors to estimate odour release values is popular for diffuse sources, where measurement is not straightforward (or routinely required for regulatory control purposes). Draft H4 Part 2 recognises that odour emission factors have been derived for only a limited range of activities, and the majority of these (falling within PPC regulation) are for the intensive livestock industry. A comprehensive review of ammonia emissions factors for intensive livestock rearing is given in Project 1 of this cluster.

Beyond PPC, there are data available for the wastewater industry. Odour formation in sewer networks and the sources of odours in wastewater treatment are discussed in detail in Stuetz & Frechen (2001). Modelling of odour formation in sewer networks is traditionally performed using empirical models for H₂S formation (Hvitved-Jacobsen and Vollertson 2000).

* But not necessarily to all the requirements of method BS EN 13725, as this publication preceded that standard.

Currently, there is no openly available database on odour emission factors across industry types specific to the UK. Many industries or companies hold information on emission rates for their own processes, much of it subject to commercial confidentiality and not representative across a spectrum of operating conditions or weather patterns.

There is a need to improve the accessibility, robustness and use of data pertaining to odour emission rate estimates. An analogy may be made with the Defra Emission Factor Database (EFD) for the Air Quality Strategy priority pollutants. This is a centralised source of emission estimates for different processes, with quality ratings and/or traceability of the source of the estimates.

A tool/resource such as this is in place in Germany (Lohmeyer *et al.* 2004a), funded by the State of Baden-Wurtemberg Ministry of the Environment and Transport. GERDA is a computer database which contains a bank of emission factors (covering biological solid waste composting plants, repair paint shops, smokehouses, sewage treatment plants and foundries). The spreadsheet contains procedures to estimate odour emission rates for each part of a site, taking account of factors such as meteorology and operating conditions. GERDA is stated to be a screening tool, which would imply that more site-specific emission factors may be required for detailed assessment.

Uncertainty in the emissions factors

There are two aspects to the uncertainty associated with an emissions factor. There are the component uncertainties from the unit odour release rate (e.g. for an area release $X_{ou_E} \text{ m}^{-2} \text{ s}^{-1}$) and from the scaling factor (e.g. temperature, number of pigs or birds, throughput tonnage or volume, etc.)

The accuracy of the scaling factor and its applicability to the site/situation in question is an issue of “portability” of the emission factor (i.e. how applicable is it to the scenario in question) and this was covered in Sections 4.3.1 and 4.3.2. Here we are only concerned with the uncertainty of the unit emissions factor itself. It is clearly important: according to Yang and Hobson (2000), odour emission rates from sewage treatment works can vary by over two orders of magnitude for a given process depending on the condition of the flow.

Emissions factors mass balance calculations are applied most commonly to diffuse sources and fugitive sources that are difficult to monitor. The Draft H4 approach - using modelling to back-calculate permissible ELVs - is not applicable to emissions factors and only limited comments have been made for the sake of completeness as it is recognised that the Environment Agency will, from time to time, receive EIAs that include odour impact predictions using emissions factors.

Bull (2004) compared Environment Agency emissions factors (Environment Agency 2003) for intensive poultry rearing with other published values and with measurements and found differences of up to an order of magnitude. Similar, if not greater variations were said to exist for published odour emissions rates for landfill sites and wastewater treatment plant.

The wastewater industry has also published some odour emission rates (WRc 1998). Yang and Hobson (2000) have suggested that the literature values of

odour emission rates have an uncertainty in the region of two orders of magnitude or more, because the degree of septicity as measured by the odour potential or sulphide concentration is never stated, nor are the precise details of the process stated (e.g. emission rates from a weir are proportional to the weir height). Their view was that establishing credible emission rates is potentially the weakest link by far in the quantitative assessment of odour impacts.

In searching for published values of odour emission factors, care should be taken that the emission rates have been obtained using the same standards of dynamic olfactometry. In the UK, the Draft H4 benchmarks are expressed in terms of European odour units (ou_E). Australia has recently developed jointly with New Zealand its own standard for odour measurement, which has used the CEN draft EN13725 as a starting point. Prior to this, the different states in Australia used very individual approaches to odour measurement, and crude conversion factors between the odour units as defined by various states and the ou_E are available (Environmental Protection Authority New South Wales 1995). Particularly noteworthy is that $2\ ou_E\ m^{-3}$ according to the Dutch NVN2820 standard is equivalent to $1\ ou_E\ m^{-3}$ according to CEN draft EN13725.

As with measurement, the emission factor can be of odour itself, or it can be of a specific compound(s) which can then be converted to odour-equivalents using published ODT values. For example, the US EPA method for estimating loading losses of gasoline vapours at oil refineries using emissions factors has a quoted uncertainty of $\pm 30\%$ (for predicting total VOCs). To this needs to be added the uncertainty associated with using a surrogate compound (covered previously in Section 4.2.2.1).

4.4.2.3 Uncertainties from odour release estimates derived from analyses of bulk materials

Liquor from open tanks may be sampled and the Odour Potential (OP) measured using published WRc methods. WRc has developed a Sewage Treatment Odour Prediction (STOP) model (Yang and Hobson 1998). The STOP model takes the odour potential of the waste and the physical parameters of the process unit to estimate odour emissions. Similarly, the CH₂M HILL Interceptor Odor Model can be used to estimate liquid and vapour phase H₂S concentrations at various locations throughout the sewage collection system and waste water treatment plant (Witherspoon *et al.* 2000).

It is not known, however, if any uncertainty estimates/margins of error are assigned to the outputs of these emissions models. In any case, these approaches do not really fall within the intended application of the Draft H4, using modelling to back-calculate permissible ELVs.

4.4.2.4 Uncertainties from odour release estimates derived from calculations

However, for the sake of completeness we have included some limited comments on other sources, as it is recognised that the Environment Agency

will, from time to time, receive EIAs that include odour impact predictions from diffuse (usually area) sources.

As with emissions factors, mass balance calculations are applied most commonly to diffuse sources and fugitive sources that are difficult to monitor. For similar reasons, the Draft H4 approach using modelling to back-calculate permissible ELVs is not applicable to mass balance calculations. The time and resource constraints of this project have prevented any detailed investigation into the uncertainties of this approach for estimating odour releases.

4.5 Summary of the uncertainties in the odour release rate value

The aim of this study is not so much to get a precise estimate of the overall uncertainty, but rather to investigate the relative importance of the main component uncertainties. It should be possible to identify, for example, whether the dispersion modelling has a much more significant component uncertainty than emissions quantification, or assignment to different bands of unpleasantness. Chapter 4 of this research report has described the uncertainties associated with the odour release value that is used as input to the next stage, atmospheric dispersion modelling. It is not necessary to quantify in detail all the component uncertainties. By convention they can be considered insignificant if less than 10% of the largest uncertainty.

There are some component uncertainties that can be quantified, and will apply to most situations as long as there is compliance with standard methods. Other component uncertainties are entirely dependent on the process, installation or application of the technique and no general uncertainty estimate can be provided. In these instances, practitioners must make themselves aware of the possible importance and use their expert professional knowledge to form a view on the likely significance.

There are, therefore, a number of choices that a practitioner needs to make at this phase of the Draft H4 assessment methodology:

- Will the release values be based on measurements or estimations?
- Will the release value be based on odour directly ($\text{ou}_E \text{m}^{-3}$), or on a specific surrogate compound, (e.g. $\text{mg H}_2\text{S m}^{-3}$) followed by conversion to $\text{ou}_E \text{m}^{-3}$ using the ODT?
- How many samples will be taken for analysis?
- What minimum level of QA/QC will be stipulated for the organisations carrying out the sampling and analysis?

These choices will have a fundamental effect on the uncertainty of the resultant odour release value and will also contribute to the overall uncertainty of the exposure prediction.

The main focus of this study is on controlled point-source releases of odour, because this is the main application of the Draft H4 modelling approach to back-calculate permissible ELVs. Figure 4.6 shows a summary of the best estimates

of component uncertainties in the release rate value (obtained by measurements) for a controlled point-source.

The Figure show clearly that the use of a specific surrogate compound followed by conversion to $ou_E m^{-3}$ using the ODT adds another layer of uncertainty to this phase, compared to measuring odour directly as $ou_E m^{-3}$. Simple inspection of these component uncertainties suggests that the uncertainty in this conversion can be the dominant factor affecting the uncertainty of the release value that goes into the atmospheric dispersion model. It is recommended that the Environment Agency makes clear in its guidance that it prefers odour release rates to be obtained directly from $ou_E m^{-3}$, and it does not favour the use of a specific surrogate compound followed by conversion to $ou_E m^{-3}$ using the ODT.

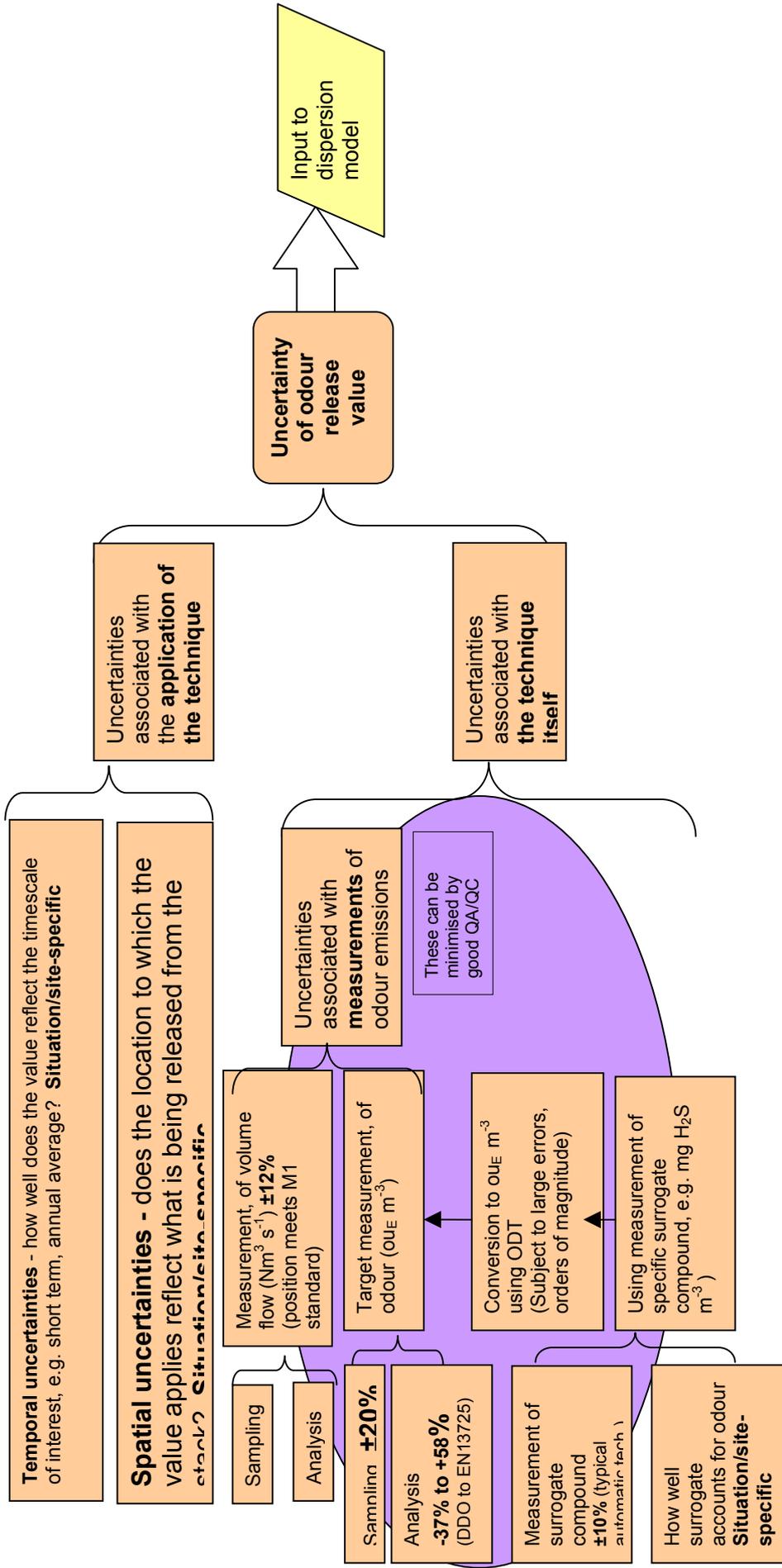


Figure 4.6 Summary of the best estimates of component uncertainties in the release rate value for a controlled point-source

5 Phase II – Modelling Odour Exposure

Step 3 needs to deal with uncertainties which can arise in the way the odour source strength, and its release characteristics (Section 5.10), are input to the dispersion model. The odour emission rate needs to be quantified only for installations if the risk of 'causing offence' is moderate or high. The discussion relates therefore only to installations for which the risk of causing offence is moderate or high, such that dispersion modelling is to be undertaken. Dispersion modelling for installations which pose only a low risk of causing offence would be a disproportionate use of resources. This is discussed further in Section 5.3.

Figure 5.1 depicts the various inputs to the dispersion model which contribute to the overall uncertainty in the model prediction. An attempt at assigning typical values to each component uncertainty is made later in Figure 5.2, on the basis of the discussions presented below in Chapter 5.

5.1 Atmospheric dispersion modelling

The link between the odour emitted at source (Step 3) and the quantity arriving at a relevant receptor (Step 5) is typically provided by dispersion models (Step 4). Environmental models are generally representations of complex systems about which there is incomplete knowledge. Quantifying uncertainty is essential for the appropriate interpretation of model outputs, particularly when assessing against environmental standards.

There is a fundamental difference between the requirements of modelling odour compared to modelling most other air pollutants. Odours can be perceived over a few seconds, whereas current dispersion models are designed principally to calculate 'ensemble mean' concentrations, typically hourly mean concentrations as the shortest averaging period for setting short-term air quality standards. The effects of odour fluctuations about the 'ensemble mean' can be modelled explicitly (see Section 5.12) or it can be incorporated empirically into statistical relationships between 'ensemble mean' concentrations (e.g. 98th percentile of hourly means) and community impacts. There is uncertainty in transposing from the hourly – average basis of modelling to the 'few seconds' over which actual odour events occur.

A research study has been carried out into uncertainty and model sensitivity analysis for the ADMS model developed by Cambridge Environmental Consultants Ltd (CERC). The study was not concerned with matters specific to odour prediction (Environment Agency 2002d) but it is recognised that such analyses are computationally intensive.

This review is not concerned with the uncertainties associated with screening techniques such as the Warren Spring d_{max} calculation or the H1 screening tool.

Rather, computerised mathematical atmospheric dispersion models are the focus of this review.

There is a need to be clear on what units the modelling uncertainty is expressed in. If an odour modelling assessment is being carried out according to the Draft H4 methodology, the exposure at a particular receptor will be predicted as the 98th percentile of hourly mean concentrations over a year. This will have units of $\text{ou}_E \text{m}^{-3}$. Any estimate of uncertainty on this prediction therefore needs to be expressed in $\text{ou}_E \text{m}^{-3}$, or as \pm a percentage of the 98th percentile prediction.

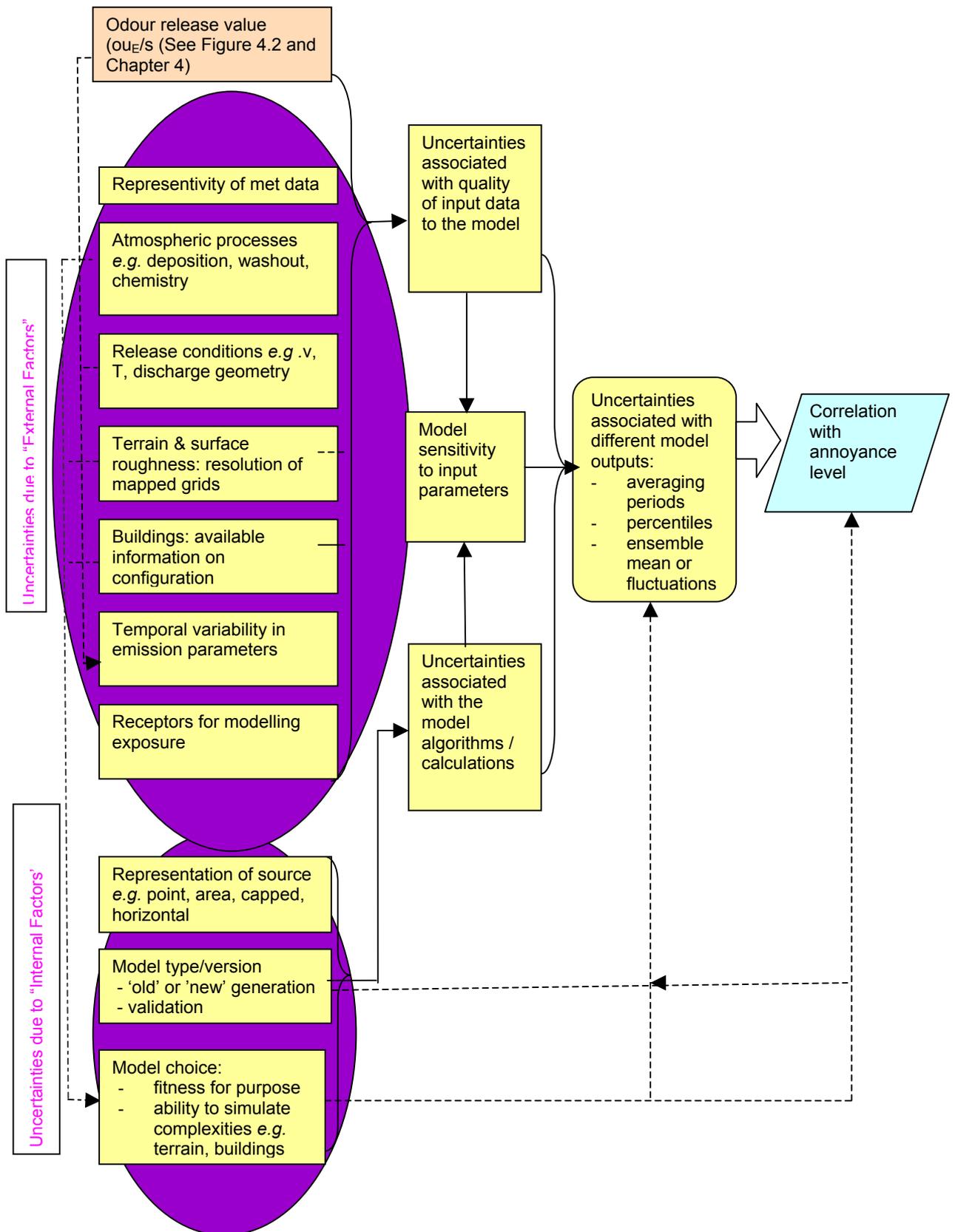


Figure 5.1 Summary of the component uncertainties in the modelled exposure value

Royal Meteorological Society guidelines 1995 (ADMLC 2004) promoting best practice in the use of dispersion models require that:

- uncertainties in model input parameters be quantified;
- the sensitivity of the model predictions to these input data be investigated;
- the inherent variability of dispersion behaviour be addressed.

The guidelines also require that an appreciation of the variation between the results of different dispersion models be demonstrated. **It is recommended that consideration be given to incorporating these guidelines into Draft H4, as a requirement of odour modelling assessments submitted to the Environment Agency.**

ADMLC (2004) provides detailed guidance on addressing sensitivity, uncertainty and variability of results obtained from using dispersion models. ADMLC (2004) also provides a useful definition of these terms, which is reproduced here in Table 5.1.

Table 5.1 Information from ADMLC (2004): Sensitivity, Uncertainty and Variability

	Definition	Statistic	Working
Sensitivity	The <i>differential</i> of model output by model input	$S = dy/dx$	An assessment may be considered sensitive to a model input parameter if varying the parameter value within a given range alters the conclusion of the assessment i.e. whether the odour benchmarks is breached or not
Uncertainty	The change in model output for a <i>plausible change</i> in model input	$S = dy/dx \times dx$	Uncertainty may be expressed by reporting a range of model results rather than a single number. For example, by running the model several times for given ranges of key input parameters, then reporting the mean \pm twice the standard deviation
Variability	That which cannot be reduced by further experiment		Some degree of variability is implicit in attempting to use a model of a natural system. For example, year on year variations in meteorology will affect the model output. In this case, variability can be expressed by using several years of meteorological data and reporting the mean \pm twice the standard deviation.

5.2 Ways of reporting modelling uncertainties

The uncertainty associated with the prediction from an individual model needs to be derived from validation exercises i.e. comparison of model outputs with field measurements. Hall *et al.* (2000a) concluded that not enough validation was available for current dispersion models.

Odour assessments require high percentiles to be modelled (e.g. the 98th percentile in line with Draft H4), but dispersion models are normally regarded as being most reliable when predicting long-term averages (e.g. annual means), and less so for shorter exposure periods and high percentiles. The level of uncertainty is therefore dependent on the averaging time.

The range of reported uncertainties for some model parameters in one example of an odour modelling study (Fraser 2005) is shown in Table 5.2. Options for the way terrain and buildings could be defined increased the model uncertainty significantly.

Table 5.2 Model uncertainty estimates for 98th percentile of hourly average predictions– expressed as percentages (Fraser 2005)

	Terrain panorama	Terrain profile	Met data	Surface roughness	Building effect	Receptor height
Range	180%	210%	66%	10%	300%	50%

These uncertainty estimates did not allow for process variation (Steps 1 to 3) or the number of calm periods (which are ignored by the ADMS model). The figures should be interpreted in the following manner: the presence of terrain meant that the predicted 98th percentile of hourly average concentrations could be about a factor of 2 higher or lower than the actual exposure at a chosen receptor location. Uncertainty over the representivity of the meteorological data was likely to lead to predicted concentrations which were $\pm 33\%$ of the actual exposure. The presence of buildings meant that predicted concentration could be about a factor of 3 higher or lower than the actual exposure.

Table 5.2 gives an example of the way in which model uncertainty could be expressed; an alternative is shown in Table 5.4, where ratios are used instead of percentages. **Consideration could be given to stipulating one or other format in Draft H4, for the sake of consistency.**

ADMLC guidance highlights the need for QA/QC of the emissions data (Phase I), topographical data (Section 5.15) and the meteorological data (Section 5.14). It advises on the collation of local meteorological data where time allows, rather than the dependence on nearest observing station, an approach which is currently prevalent for assessments submitted to the Environment Agency. Bull (2004) recognised that a wide range of results can be obtained from dispersion models, if uncertainties in the input data are examined.

There is no agreed, generic approach for addressing model uncertainty for regulatory purposes. Using an Environment Agency-suggested approach (Shi and Ng 2002) for elevated point sources, uncertainty has been loosely defined

as the degree to which the set of model predictions differs from the equivalent set of observations. At this workshop, the *model headroom* was identified as an important parameter. For odour modelling, the headroom would be the gap between the benchmark and the predicted odour concentration, divided by the benchmark. For example, if the benchmark for a particular odour is taken to be $3 \text{ ou}_E \text{ m}^{-3}$ as the 98th percentile, and the model prediction is $1 \text{ ou}_E/\text{m}^3$, the gap is $2 \text{ ou}_E \text{ m}^{-3}$, and the headroom is 0.666 (i.e. 2 divided by 3).

The smaller the headroom, the greater is the confidence required of the model predictions, to reduce the overall risk. Conversely, if the headroom is large, less confidence in the model predictions could suffice. The confidence placed on the model prediction for high percentiles depends on whether terrain and/or buildings are included, number of years of meteorological data, etc. Table 5.3 below summarises the interpretation of model headroom and model confidence levels developed at the Environment Agency workshop.

Table 5.3 A definition of ‘Low’, ‘Medium’ and ‘High’ for model headroom and model confidence levels for high percentile of hourly mean values

	High/Large	Medium	Low/Small
Model headroom	> 0.8	0.3 – 0.8	<0.3
Model Confidence level	Flat terrain, no significant buildings, 5 years of representative met data, no coastal effects	Single rectangular buildings or complex terrain, or between 3 and 5 years of met data or predictions for specific receptors	Groups of buildings, or irregular/non-solid buildings, or non-representative met data or any combination of those in the ‘medium’ box

This summary recognises that the modelling predictions for a particular point in space are less reliable than identifying the maxima over a grid of receptors. It also recognises that combining complex effect (e.g. buildings with terrain) begins to rack up the uncertainty rapidly.

This ‘headroom’ approach is more suited to the assessment of pollutants which have statutory air quality standards set as fixed values. It is less useful for odour, for which the benchmarks are set as bands depending on offensiveness or character, as is the case in Draft H4, and where the annoyance potential is highly subjective (Steps 6 to 8).

ADMFC has addressed in detail the methods for undertaking uncertainty analyses (2001). The discussions covered the stages in an uncertainty analysis, Monte Carlo techniques, Bayesian techniques and the derivation of statistical parameters to describe relationships between model inputs and outputs. The impracticalities of running a very large number of scenarios were highlighted. However, the discussions did not cover dispersion fluctuations, which are of critical importance for odour modelling. Modelling of short timescale fluctuations is covered in more detail in Section 5.12.

A 'factor of 2' difference in predicted concentration is commonly quoted as a tolerable limit of accuracy for model inter-comparisons or model validation. It is based mainly on practical experience but has limited theoretical justification. An example summary of the accuracy of model predictions for stack emissions is presented in Stuetz and Frechen (2001), and reproduced below in Table 5.4 . This applies only to the accuracy of the dispersion model itself. When there is additional uncertainty over emission rates, the possibility of errors is likely to be greater. It is therefore necessary to regard model predictions as an *indication* of expected concentrations, rather than an absolute prediction. According to this summary for this particular example, the greatest uncertainty was in the prediction of the maximum hourly concentration, where it could be underestimated by 90% or over-predicted by a factor of 10. The greatest certainty was in the prediction of the value for the peak annual mean concentration (but not its location), which could be achieved with an accuracy of $\pm 25\%$. The factors are largely consistent with the percentages reported in Table 5.2.

Table 5.4 From Stuetz and Frechen (2001): Accuracy of Model Predictions

Application	Ratio of predicted to observed concentration
Peak annual concentration (not its location). Flat terrain, steady atmospheric conditions	0.75 – 1.25
Specific hour and receptor point, flat terrain	0.2 – 2.0
Maximum hourly concentration (100 th percentile) from a continuous source, short range, flat terrain	0.1 – 10
Annual average concentration (approx. 50 th percentile) at a specific point, short range, flat terrain	0.5 – 2.0

5.3 Is detailed modelling appropriate?

It is important that dispersion modelling is not misused. The New Zealand Good Practice Guide (Ministry for the Environment 2003) advises to:

- Only use odour dispersion modelling for new activities where the predominant odour effect is due to normal process discharges that are continuous or semi-continuous and reliable odour emission rates are available;
- not use odour dispersion modelling to try to prove the *absence* of an adverse effect when community data can be collected, or are available to demonstrate the current level of effect;
- not use dynamic dilution olfactometry (DDO) measurement and dispersion modelling to investigate potential *acute* effects of odour discharge.

This recognises the reliance of the modelling on robust emission data (Phase I). It also recognises the strength of dose-effect data compared to modelling predictions (Phase III). However, in the UK, dispersion modelling of odours is often carried out for cases which do not meet these stringent criteria.

Consideration should be given to incorporating this guidance into Draft H4.

However, the Environment Agency view may be that if short term emissions data and modelling are accurate enough, then it should be possible to investigate the potential for short-term events (e.g. failure of abatement equipment) to cause acute impacts. Special care is needed to investigate occasional acute events, when routine levels of uncertainty will not be adequate for such cases.

5.4 'Old-Generation' models: COMPLEX1 and LTDF

'Old-generation' models have their origins in Gaussian plumes, wherein the concentration is assumed to be normally distributed in space about the plume centre-line, and the plume spread is described by the dispersion parameter σ_x and σ_y (the root mean square plume widths in the crosswind and vertical directions, which vary with averaging time). Gaussian plume models have some short-comings, the most important of which is that they do not represent the structure of the atmospheric boundary layer in the detail which is possible in more recently developed similarity-theory based models. New-generation models recognise that convection is not a symmetrical process, and this can lead to non-Gaussian concentration profiles, especially in unstable conditions. The 'old-generation' models are characterised by their use of 'Pasquill stability categories' to describe the state of the atmospheric boundary layer. In contrast, 'new-generation' models describe atmospheric stability using the Monin–Obukhov length scale (Middleton and Thomson 2002).

It is important to note that both 'old' and 'new' generation models (and indeed Lagrangian and Eulerian models) are essentially statistical in character. They do not predict the actual concentration field in a given situation, but rather the 'ensemble mean' that would be expected if one could repeat the scenario many times with essentially identical external conditions. Such repetitions would differ by random turbulent fluctuations (Section 5.12).

It has been stated in some of the literature (and colloquially) that the Dutch dose-effect study used the COMPLEX1 dispersion model (Bull, personal communication, 2006). However, on investigation, it is concluded that the Dutch dose-effect relationship used LTDF modelling (Darren Hall, Entec, personal communication, 2006). It has not been possible to obtain the original Dutch modelling data; it would have been helpful for the purposes of Project 3 if details of the LTDF odour modelling were available for examination. Of particular concern is the way in which the source strength was defined in the Dutch modelling.

The Environment Agency's Report P4-095/TR (2002b) also states that the Dutch regulatory model 'Lange Termijn Frequentie Distributie' (LTDF) (VROM 1984), was used to calculate exposure for the Dutch dose-effect studies for pig

production units (Bongers *et al.* 2001). This dose-effect relationship has been adopted by Draft H4 as a tool for predicting annoyance levels in a population. LTDF was the Dutch national regulatory model until 2000, when it was replaced by a new modern Gaussian model. A study re-examining the original annoyance survey results in 2000 has again cited the dispersion model as being the Gaussian LTDF model (Werkgroep Verspreiding Luchtverontreiniging 1983).

Although many model inter-comparison studies between 'old' and 'new' generation models have been reported, none have included LTDF. Any modelling results which the Environment Agency receives as part of an odour assessment in the future will almost certainly be based on ADMS or AERMOD PRIME. **Although outside the scope of Project 3, it would be useful (perhaps as part of another research project) to attempt to identify any systematic differences in predictions between LTDF and both of the two 'new-generation' models, for cases which are relevant to odour assessment e.g. high percentiles, and the relevant range of stack heights.** This would identify any bias-type (systemic) differences between the modelling that established the dose-response relationship, and modelling results which the Environment Agency is likely to receive now.

COMPLEX1 was used for the modelling carried out for the planning appeal against refusal of consent for a wastewater treatment works at Newbiggin-by-the-Sea – the decision of the Inspector (issued in 1993) set a precedent for hourly modelling of the 98th percentile, against a benchmark of 5 ou_E m⁻³ which was adopted by the UK odour modelling community and industry for many years after the public inquiry.

COMPLEX1 is a module for complex terrain incorporated into the ISCST3 model (US EPA 1995). When complex terrain is encountered, the model determines the elevation change for the leading hill and raises the plume height by half of that amount. This is done in an attempt to allow the plume to respond to the underlying terrain, while maintaining conservatism. It was recognised by the US EPA that such ISC terrain predictions were very poor, as the model tended to over-predict significantly, by at least a factor of 2 (USEPA 1999). This partly underpinned the development of AERMOD. On this basis, it may be assumed that the use of COMPLEX1 in areas of flat terrain would revert the modelling to the original ISCST algorithms.

5.5 'New-Generation' models: ADMS and AERMOD

'New-generation' dispersion models are very versatile in terms of being able to incorporate changes in odour emission rates on an hourly/daily/monthly/seasonal basis. The principal difficulty lies in measuring, estimating or predicting the odour emission rate for the various stages in a production cycle/operating regime (Steps 1 and 2). This is particularly important for odour assessments where there can often be significant diurnal variation in the odour source strength.

Currently, applications for authorisation are most likely to be based on modelling using ADMS or AERMOD. The relative performance of these models is of concern, and has been investigated in detail by Hall *et al* (2000b) for a range of relatively simple scenarios (though the PRIME algorithms for building downwash effects were not included in AERMOD at the time of that study). A critical feature of inter-comparisons appears to be the handling of meteorological data inputs to the models, especially in the relationships between Monin-Obukhov length scale inputs.

It is clear that there are significant, case-dependent differences between the two 'new-generation' models, AERMOD and ADMS. The former was developed by the US EPA, and the latter was developed by CERC. Hall *et al.* (2000a) concluded that the greatest differences between the two 'new-generation' models ADMS and AERMOD is probably in the secondary procedures to deal with factors such as plume rise, building entrainment and terrain. There is some justification therefore for assuming that US EPA models (e.g. ISC and AERMOD) should be more akin to each other than the two 'new-generation' models developed by different organisations. This was been found to be the case between AERMOD and ISC in relation to the way that they treat buildings. However, AERMOD PRIME has now replaced AERMOD, and has enhanced methods for treating buildings (Section 5.7).

The Draft H4 Indicative Odour Exposure Standards were derived from comparison of the output of an 'old-generation' model (LTDF), with annoyance surveys. Draft H4 states that *'If using newer models to compare the actual performance against the benchmark, it is possible that the installation-specific results may show a numerically higher result than would have been the case with the use of an older-generation model'*. This recognises that an 'old-generation' model could give *higher* predicted ground level concentrations than a 'new-generation' models, for the same model input parameters. If so, then the Indicative Odour Exposure Standard would need to be *lowered* if 'new-generation' models are being used to predict the impact of the same odour-emitting installation, and assuming everything else remains constant. Conversely, the 'new-generation' model would allow the plant a *higher* odour emission rate to still achieve the same predicted odour concentration (as $ou_E m^{-3}$) at any particular receptor.

Hall *et al* (2000a, 2000b) implied that there is less consensus between the 'new-generation' models (ADMS and AERMOD) than between the older models (R91 and ISC). For Project 3, this assumption could have been useful if it had been possible to affirm that LTDF was typical of an 'old-generation' model such as ISC. However, in the absence of any published inter-comparisons between old-generation models which included LTDF, it is not possible to *assume* that the results of LTDF modelling were typical of, or similar to, the results of ISC or R91.

The major effort in model development, besides extending the range of application, has tended towards minimising systematic errors in calculation and in trying to define the degree of uncertainty associated with model calculations. It is common for substantial differences to exist in nominally identical calculations between different *versions* of the same model. Model developers can argue that the current version of any model is the best validated and best

fits the field data. Therefore, there may be little point in now discussing older model variants.

ADMS-4 is expected to be released by CERC in late 2006. The most up-to-date release for ADMS at the time of writing (April 2006) is ADMS-3.3. This was released in late July/early August 2005, and replaced ADMS-3.2. At the time that the Environment Agency let the contract for the project P4-120/2 Ref. 13933, the most up-to-date version was ADMS-3.1. Concern has been expressed in the past over significant changes in the predictions of different versions of ADMS.

It is understood that Air Quality Monitoring and Assessment Unit (AQMAU) routinely investigate the performance of different version on their release. It is likely that there are significant differences between ADMS-2, ADMS-3.2 and 3.3 in the way building downwash is treated. Moreover, CERC state that ADMS-3.3 corrects an error in ADMS-3.2 concerning the calculation of percentiles. ADMS3 calculates percentiles by linear interpolation of a probability distribution function (PDF) generated from the predicted hourly concentrations at each receptor. This raises concerns over the calculation of percentiles (notably the 98th percentile, of relevance to Project 3) in published inter-comparison studies. It also highlights the more robust approach of identifying the number of exceedances of an hourly mean benchmark, rather than depending on the software or post-processor to calculate percentiles automatically.

A recent, published inter-comparison of 'new-generation' models by AQMAU (Sidle *et al.* 2004) included ADMS3.1, AERMOD and AERMOD PRIME. This study concluded that there was greater agreement between the models for the 98th percentile than higher percentiles. When buildings were included, it was observed that the models showed very different sensitivities to the wind angle and few trends could be discerned. The results of terrain modelling were very similar between the US EPA models, but very different to the ADMS3.1 results. **The authors concluded that for regulatory purposes, it is advisable to make use of two models to increase confidence in the model predictions.**

5.6 Model inter-comparison studies

Such studies are well-documented, but are in danger of dating very quickly as a result of newer versions being released to replace older versions. For example, AERMOD PRIME was promulgated recently by the US EPA and this is now the *de facto* standard for AERMOD. Therefore, it is unlikely that the Environment Agency will any longer receive an application for an odour-emitting process which uses AERMOD without the PRIME algorithms, if buildings are present. The Environment Agency is even less likely to receive any applications based on ISCST3.

Key features of published inter-comparison results are summarised in this review where they could be relevant to the types of situation encountered for odour-emitting installations in the UK. For example, studies describing the

modelling of very tall stacks (greater than 75 m) are excluded^{*}, and the results of short averaging period/high percentile (i.e. 98th or greater) calculations only are of primary interest. It is unfortunate that no published inter-comparison studies have been identified which included LTDF.

Lee *et al.* (1999) compared the results of AERMOD and ISCST3 for a refinery, predicting the 3-hour high-second high SO₂ value (i.e. the second highest, 3-hour mean concentration, which is equivalent to the 99.9th percentile). This paper draws attention to an important issue which has relevance for odour modelling, namely that ISCST3 uses the Pasquill dispersion curves which apply to a 'sampling time of about 3 minutes', whereas AERMOD predicts 1-hour averages by design. There were 22 sources on site (stack heights ranging from 13 to 41 m), significant downwash effects, and flat terrain. The AERMOD and ISCST3 predictions for the 3-hour, high-second high values were within 13% of each other, which is surprisingly good agreement for two dispersion models for a high percentile calculation; it may be partly due to the results for the 22 sources being aggregated, and the errors cancelling each other out to a large extent. No systematic trend was observed, one model giving higher results than the other depending on which of four meteorological years was being modelled.

Hall *et al.* (2000a) reviewed inter-comparison studies of ISC, R91, ADMS and AERMOD published prior to September 2000. It is unclear which version of ADMS was used in each of the studies, and this could be a significant confounding factor. They stated:

'Only ten studies were found directly comparing either ADMS and AERMOD with each other or with older models....The effects of building entrainment and topography in particular have been little studied. Little attention has been given to variant versions of model issued over time and any differences between them. A critical feature of inter-comparisons appears to be the handling of meteorological data inputs to the models, especially in the relationship between Monin-Obukhov length scale based inputs and Pasquill/Gifford stability categories'.

A few relevant, simplified findings of the review carried out by Hall *et al.* and other inter-comparison studies are reproduced below:

- Hall *et al.* (2000a) concluded that differences between 'new' and 'old-generation' models are generally greater in stable and unstable boundary layers, than in neutral conditions. The 'new-generation' models tend to produce more rapid vertical dispersion in unstable flows, and reduced vertical dispersion in stable flows.
- A study by Jones *et al.* (1995) found that maximum 1-hour mean concentration (i.e. the 100th percentile) predicted by R91 and ADMS1 tended to be different by a factor of between 2 and 3, ADMS-1 generally predicting the lower concentrations. The stack heights in the Jones *et al.*

^{*} Previous discussions with Chris Sidle (Environment Agency) concluded that stacks exceeding 75 m were very rare for installations of interest to the Agency with respect to odour impact.

(1995) study were in the range 10 – 70 m, and three building shapes were modelled

- Maul *et al* (1996) compared two versions of R91 with ADMS and ISCST2, for stack heights in the range 20 – 200 m – building effects were not included. Maximum 1-hour mean concentrations (i.e. 100th percentiles) varied by up to a factor of 4 between models. ADMS produced significantly more rapid vertical dispersion in convective conditions and reduced dispersion in stable conditions. This considerably altered the distances at which the maximum ground level concentrations occurred.
- Harvey (1998) has presented a comparison of high percentile predictions for AERMOD and ADMS, for installations with stack heights of 70 m (with a low building), 50 m (with a low building) and 55 m (no building). Some of these data are summarised in Table 5.5. The 98th percentile predictions are important for the purposes of identifying any systematic differences between ADMS and AERMOD in relation to odour assessment in line with Draft H4.

Table 5.5 Some data from Harvey (1998) : Percentiles of predicted hourly-average concentrations over a year.

Percentile	Model	70 m (with low building),	50 m (with low building)	55 m (no building).
100th	ADMS	142	267	222
	AERMOD	31	234	62
99.8th	ADMS	58	245	41
	AERMOD	17	122	32
99.5th	ADMS	25	204	35
	AERMOD	15	101	27
99th	ADMS	15	190	30
	AERMOD	13	88	21
98th	ADMS	11	151	23
	AERMOD	10	72	12

These results show that AERMOD generally gives lower predictions than ADMS in the absence of tall buildings; they can be lower by a factor of 4 for the very high percentile (100th), but more typically a factor of 2 for the 98th percentile.

- Paine *et al.* (1998) compared AERMOD and ISCST3 against field data for tall stacks. Building downwash effects are likely to have been insignificant. They found that both models tended to under-estimate lower concentrations, but the performance improved for the higher concentrations (which would be more relevant for an odour assessment).
- Hanna *et al.* (1999). This study involved a validation and comparison of ISC3, AERMOD and ADMS3 against 5 field datasets. A factor of 8 difference for the 100th percentile was possible between the models when severe building entrainment was present. It is noteworthy in this respect that AERMOD did not at that time incorporate the PRME algorithms. With low buildings and no obstacles, ISC tended to over-predict the 100th percentile compared to field data, whilst ADMS and

AERMOD tended to under-predict. The study concluded that the ‘new-generation’ models performed considerably better than ISC3, and the performance of ADMS3 and AERMOD was broadly similar.

- Garrison and Sherwell (1997) compared AERMOD and COMPLEX1. They found that AERMOD represented an improvement on COMPLEX1 because it responded better to continuously changing meteorological conditions. It is likely that terrain effects dominated over any building downwash effects.
- When terrain effects were investigated by Peters *et al* (1999), ISC incorporated the COMPLEX1 algorithms. They found that AERMOD predicted consistently lower concentrations (by more than a factor of 2 for most cases) than ISC/COMPLEX1. Again, ‘new-generation’ gave lower result than ‘old-generation’.
- McIntyre (2000) presents comparisons of the 98th percentile of hourly mean predictions from ISC and AERMOD. ISC consistently predicted higher values than AERMOD; these are summarised in Table 5.6 below. ISC over-predicted compared to AERMOD by factors in the range 1.2 – 3.2.

Table 5.6 From McIntyre (2000): 98th percentile predictions of hourly means

Model	Receptor 1	Receptor 2	Receptor 3	Receptor 4
<i>Existing Case</i>				
AERMOD	7.0	3.4	8.1	3.8
ISC	10.6	7.5	9.9	7.2
<i>Future Case</i>				
AERMOD	4.3	2.5	2.7	1.5
ISC	6.9	6.1	6.6	4.8

Following on from their review, Hall *et al.* (2000b) carried out their own model inter-comparison of AERMOD3.2.2, ADMS3.0 and ISCST3.2.2; all versions which were released in 1999. Hall *et al.* (2000b) used a protocol of test cases to examine the behaviour of the three dispersion models. The test cases cannot encompass all possible uses of the models but attempted to cover as wide a range of practical cases as possible. It is also noteworthy that AERMOD PRIME is now the promulgated version of AERMOD, and this has different building downwash treatments from the version of AERMOD investigated by Hall *et al.* in October 2000.

A 40 m stack was included in the test protocol (and a ground level source for certain scenarios), with buildings of 25 m and 35 m, for both buoyant and non-

buoyant releases. The terrain in the vicinity of Porton Down was modelled, with various orientations for the source relative to the terrain. Lyneham 1995 meteorological data were used. Findings relevant to odour impact assessments are reproduced in Table 5.7, which relate to specific 1-hour meteorological conditions.

These results show that the 'new-generation' models are more likely to give lower predictions for maximum concentrations under user-defined short-term meteorological conditions (as compared to ISC). However, for unstable conditions, ADMS predicts the highest concentrations. In terms of the distance from source to the maximum concentration, AERMOD predicted the peak at far greater distances under stable conditions compared to the US EPA models.

The results in Tables 5.7 and 5.8 again show that the 'new-generation' models are more likely to give lower predictions for maximum concentrations under user-defined short-term meteorological conditions (as compared to ISC). In terms of the distance to the predicted maximum, no systematic difference between the models could be identified for the test scenarios.

Table 5.7 Parts of Table 7 from Hall et al. (2000b) Normalised concentrations as $m^{-2} \times 10^6$ (no buildings)

Scenario	Model	Neutral/low wind speed		Neutral/high wind speed		Unstable		Stable	
		Con c.	Distance to maximum (m)	Conc.	Distance to maximum (m)	Conc.	Distance to maximum (m)	Conc.	Distance to maximum (m)
Non-buoyant release	AERMOD	24.5	1000	52.4	500	38.9	300	12.1	3,000
	ADMS	24.8	700	44.3	500	74.1	200	7.9 – 9.5	1,700 – 1,400
	ISC	49.8	900	60.0	800	62.3	300	19.6	1,900
Buoyant release	AERMOD	4.7	2900	31.4	650	11.8	350	2.2	10,000
	ADMS	4.4	2100	28.4	550	22.0	300	1.5 – 2.2	3,400-4,900
	ISC	10.9	2100	27.2	1200	16.7	600	5.4	3,900

Table 5.8 Parts of Table 9 from Hall et al. (2000b) Normalised concentrations as $m^{-2} \times 10^6$ (including scenarios with buildings)

		Model	Neutral/low wind speed		Neutral/high wind speed	
			Distance to maximum (m)	Conc.	Distance to maximum (m)	Conc.
40 m stack	No building	AERMOD	1000	25	500	52
		ADMS	700	25	500	44
		ISC	900	50	800	60
40m stack	25 m high building	AERMOD	180	105	170	100
		ADMS	200	90	200	120
		ISC	110	250	120	190
40 m stack	35 m high cubic building	AERMOD	110	165	110	160
		ADMS	200	130	0	260
		ISC	110	250	120	190

Ground level	35 m high cubic building	AERMOD	110	1100	110	800
		ADMS	0	3350	0	3820
		ISC	110	1020	110	1020
40 m stack	35m high, very wide building	AERMOD	110	165	110	160
		ADMS	300	140	0	150
		ISC	110	250	120	190

Table 5.9 Parts of Table 10 from Hall et al. (2000b) High percentile results, flat terrain case, Lyneham 1995 met data Normalised concentrations as $m^{-2} \times 10^6$ (including scenarios with buildings)

	Building	Model	100 th percentile		99.9th percentile		98 th percentile	
			Conc.	Distance to maximum (m)	Conc.	Distance to maximum (m)	Conc.	Distance to maximum (m)
40 m stack, no buoyancy	No building	AERMOD	58	100	33	140	11	280
		ADMS	116	100	69	100	11	280
		ISC	45	300	40	315	13	780
40m stack, with buoyancy	No building	AERMOD	6.3	200	4.7	280	3.0	420
		ADMS	12.9	100	8.4	220	3.3	500
		ISC	7.1	1420	5.1	810	3.1	1490
40 m stack, no buoyancy	25 m building	AERMOD	101	140	79	220	28	140
		ADMS	690	1500	154	0	62	140
		ISC	144	200	90	300	35	200

5.7 Model inter-comparison results with building downwash effects

Such studies are well-documented but are almost all out-of-date, as more refined versions of most models are released which claim to have improved methods for dealing with downwash effects. Tables 5.8 and 5.9 summarise some pertinent, historical results. Without buildings, the 98th percentile results can be quite similar between ADMS and AERMOD models. However, with buildings, ADMS and AERMOD predictions can differ greatly, with ADMS predicting much higher values.

There is concern over a recently disclosed error in percentile calculation within ADMS3.2, which has been reported on release of ADMS 3.3. Moreover, there have been numerous changes in the building wake/entrainment treatments in ADMS with successive versions of the model.

ISC is closer to AERMOD which would be expected for two US EPA models containing similar downwash algorithms. In terms of the distance to the predicted maximum concentrations, ISC generally gave the peak furthest from source, with AERMOD and ADMS predicting more similar distances on most occasions.

These published result show that the general order of predictions for high percentile (98th, 99th and 100th) concentrations in the presence of building downwash effects, is as follows:

- Highest prediction: ADMS
- Lowest predictions: AERMOD

However, this trend is at variance with a more recent study carried out by BV in 2004 (for multiple controlled-stack odour emissions at a car-manufacturing plant with significant buildings), which employed ADMS3.1 and AERMOD (with PRIME algorithms). Stack heights were all 30 m. It was found that AERMOD gave higher predictions for the 98th percentile concentration at most receptor locations, by as much as a factor of 4 at the point of maximum impact, but generally about 20% higher than the ADMS3 result at specific locations of interest. Receptors were all within 2 km of the emission sources, and some were within 100 m of certain stack-building configurations.

In this respect, it would have been particularly useful to examine the LTDF modelling which established the Dutch dose-effect relationship, for the inclusion and treatment of building effects. Regrettably, no information on the technical details of the historical LTDF modelling were available for Project 3.

Just because a model claims to be able to treat buildings and terrain does not mean it works well. ADMLC (2004) highlights the uncertainties surrounding models treatment of terrain and buildings, and their sensitivity to surface roughness (often overlooked for sensitivity analysis). The entrainment of plumes in building wakes is of great practical importance and is often the

dominant factor controlling acceptable stack heights, especially for low level stacks. Hall *et al* (2000b) concluded that AERMOD, ISCST3 and ADMS all treated near-field building entrainment in quite primitive ways. For example, ADMS approximates the building form into an equivalent rectangular block. The model inter-comparison study of Hall *et al.* concluded that none of the models was effective at estimating near-field building plume downwash and entrainment. Subsequent to the work of Hall *et al.*, arguably the most robust field validation is now available for AERMOD PRIME (Peters *et al.* 1999).

It has not been possible to demonstrate any relationship between the results of LTDF and ISC during the course of Project 3. On the basis of the model inter-comparison studies which have been reviewed (Sections 5.6 and 5.7), the following approach could be considered: **more than one 'new-generation' model should be used for any odour modelling assessment. The range of results obtained from the use of multiple models needs to be inspected, to ascertain the risk of breaching the IOES.**

5.8 Choice of model

In any modelling study, the choice of model needs to be justified, with particular reference to local building configuration, surface roughness and terrain (Section 5.15). There is a need to identify any systematic error (bias), for example one model always predicting lower than another.

A variety of dispersion models, different versions of models and approaches to modelling may be used in applications for authorisation. The applicant must demonstrate that the model used is fit for purpose*. Report P4-095/TR (Environment Agency 2002b) acknowledges that:

Ideally, when using dispersion models for odour annoyance prediction, the objective must be to apply the models that were used to establish dose-effect relationships in the underlying epidemiological case studies.

The Dutch regulatory LTDF model was used for the modelling which calculated the odour concentration bands for the dose-effect relationship in the Netherlands. LTDF pre-dated the 'new-generation' models such as ADMS and AERMOD, but its similarities (if any) to ISC are not known. The results of the LTDF modelling have been largely adopted by Draft H4. Further details of this Gaussian model, and also details of the modelling scenarios, have not become available for Project 3.

Draft H4 acknowledges that the modelling of odour is still a developing field. It states:

'A range of different models have been used for odour modelling and have a number of common features, but there are differences in the way that data is dealt with between the older Gaussian model and the 'new -generation' models such as AERMOD and ADMS'.

* Environment Agency Policy Number Environment AgencyS/2007/1/1

Fisher (2001) has proposed that when results from several dispersion models are available, and assuming that each model has an acceptable level of performance (e.g. ADMS and AERMOD), the inevitable inaccuracies in each model should be accepted and both sets of prediction be used to obtain a combined estimate of the impact based on the information available. For installations which present a high risk of odour annoyance, and for which dispersion modelling is to be carried out, **consideration could be given to a requirement in Draft H4 that two appropriate dispersion models be applied to the investigation.** This forms the basis of a risk-based approach.

5.9 Treating predictions from more than one model

Deterministic modelling neglects uncertainty. This has been discussed by Fisher (2001) who suggests that the predictions from 2 models could be regarded as either end of a range. He proposes that the prediction of a model should not be described as a single number, but as the best estimate of a range of numbers, using, for example, fuzzy numbers or fuzzy sets. This uncertainty can be described in a number of ways. The uncertainty on each number can be used to derive a joint weighting of uncertainty based on statistical methods.

The treatment of uncertainty is inevitably associated with some subjective judgements. Fisher concluded that using results from more than one model has distinct advantages in decision making, if used with care. Regulatory decisions should reflect the underlying uncertainty. These potentially could lead to a better description of imperfect knowledge.

The regulator is best placed to encourage the applicant/operator, on a case-by-case basis, to submit results using more than one dispersion model, and/or to carry out community dose-effect studies. It would appear that though applicants have to demonstrate that the assessment is representative and fit for purpose, in practice no mechanism is in place for *obliging* applicants to submit an odour assessment to this level of detail and complexity.

5.10 Defining model input parameters

For dispersion modelling, data on discharge conditions (efflux velocity/momentum and heat release) are required, all of which are associated with uncertainties due to the measurement or estimation techniques. It may also be necessary to measure rate of air flow or ventilation (e.g. number of air changes per hour). These issues are addressed in Section 4.4.1.3.

5.10.1 Source location

It is not unreasonable to expect that, for the vast majority of installations, the locations of the odour source/s (stack or areas of known shape) should be known with very high precision e.g. an OS grid co-ordinate to the nearest 1 m. However, the predictions of dispersion models, if used to identify the point of

maximum odour impact, can have uncertainties of hundreds of metres. It is important to define the stack location to the nearest metre because then the *relative* locations/proximity of nearby buildings can be treated appropriately by building downwash algorithms. Area sources need to be sized and located accurately so that the *total odour* emitted (gs^{-1}) is calculated correctly. With modern GIS interfaces to dispersion models, accurate positioning of source, buildings, terrain heights, site boundaries and receptors renders negligible the uncertainties in dispersion modelling results arising from mis-positioning of these elements. The uncertainties in dispersion modelling exercises which include buildings and terrain are discussed in Sections 5.7 and 5.15 respectively.

5.10.2 Horizontal releases

Dispersion models were formulated for unobstructed vertical releases i.e. chimneys. The dispersion of pollutants from horizontal pipes lends itself more to computational fluid dynamics (CFD) modelling, and cannot readily be investigated using practical, proprietary models. Guidance from the US EPA states that an exit velocity of 0.001 ms^{-1} should be assumed for a horizontal release, in order to eliminate the momentum plume rise (J. Hill, Trinity Consultants, personal communication, January 2006). In addition, stack tip downwash in the model must be switched off, whether or not there are buildings present. Stack tip downwash calculations are inappropriate for horizontal stacks. The details of the modelling approach adopted by the applicant for modelling horizontal releases should be checked for these assumptions. It must be recognised that the uncertainty on a 98th percentile prediction for horizontal releases will be very high, but that the prediction is likely to be an overestimate. **Therefore, it is recommended that careful consideration be given to the validity of odour dispersion modelling for horizontal releases.**

5.10.3 Capped release points

Dispersion models were formulated for unobstructed vertical releases, i.e. chimneys. Guidance from the US EPA states that an exit velocity of 0.001 ms^{-1} should be assumed for a stack with a cap, in order to minimise the momentum plume rise (J. Hill, Trinity Consultants, personal communication, January 2006). In addition, stack tip downwash should be maximised because the cap will probably generate such downwash most of the time. The resulting predictions of downwind concentrations may err slightly on the high side. The details of the modelling approach adopted by the applicant for modelling stacks with caps should be checked for these assumptions. It must be recognised that the uncertainty on a 98th percentile prediction for capped releases will be very high, but that the prediction is likely to be an overestimate. Therefore, if such a case is modelled, a risk-based approach is needed: if the predicted odour concentration is low, then even with modelling uncertainty there might not be a problem.

5.10.4 Multiple sources

The results of dispersion models are inherently more uncertain in the *following order*:

- Single stack;
- Multiple stacks spaced well apart;
- Multiple stacks in close proximity;
- Single area source;
- Multiple area sources.
- Non-vertical or obstructed releases.

On this basis, if multiple area sources, non-vertical or obstructed releases are modelled, it must be recognised that there could be large uncertainties in model predictions. A risk-based approach is needed. If the predicted odour concentration is low, then even with model uncertainty there might not be a problem.

This ranking does not take account of how odours from different 'plumes' may react with each other, change their characters on mixing, etc. It is not possible with current dispersion models to account for these factors, though most 'new-generation' models (e.g. ADMS3, AERMOD PRIME) are very capable of source apportionment for conservative pollutants. They can screen out certain sources in relation to particular receptors, enabling a detailed investigation of uncertainties associated with only the significant sources. However, it is important to bear in mind that odours are not simply additive, like other conservative pollutants. Masking and synergistic effects are complex and vary for each mixture of odorant. This is discussed in detail in the Literature Review for Project 2.

Where there are multiple sources of odour and/or mixtures of chemical compounds, dispersion modelling is likely to over-predict downwind odour impacts. This is because the diluted odour mixture will be dominated by the more offensive components in the mixture, which mask the less offensive components. This over-prediction could be a significant uncertainty in the overall odour impact assessment. The New Zealand Technical Report (Ministry for the Environment 2002) considers two options to moderate the effect of this over-prediction:

- Where the odour discharges on a site can be classified according to their offensiveness (e.g. 'very offensive' and 'slightly annoying'), the 'very offensive' sources are likely to dominate the 'slightly annoying' sources (unless the latter has a very high predicted downwind concentration relative to the other group). Therefore, the groups of sources can be modelled separately.

- Where a small number of sources on a site are of much lower offensiveness than the others, then the odour emission rate determined for these least offensive odour sources could be corrected downwards by dividing the source's emission rate by suitable factors.

Consideration could be given to the incorporation of these options into future revisions of Draft H4, in order to reduce the uncertainty associated with modelling multiple odours on a single site. The guidance in Draft H4 currently is as follows:

- Where sources are spread across a large installation, it may be appropriate to consider individual or groups of sources in terms of the specific receptors that may be affected. There may also be a need to consider the effect of distance from the source to different receptors, depending on wind direction.
- Where sources are completely different in terms of their odour characteristics, it may be better to consider them separately in terms of impact on receptors.

5.10.5 Area and near-ground sources

Published/documented validation of dispersion models against field measurements for real area sources is scarce. Dispersion models were developed primary for single chimneys, but have been modified subsequently to treat other types and numbers of source. In this respect, it would have been particularly interesting to examine the LTDF modelling (which established the Dutch dose-effect relationship) for the inclusion of area and/or multiple sources.

It is not clear whether ADMS3.3 has been validated against field measurements for single area or volume industrial sources. CERC refer to the OPTEX and Duke Forest experiment, but these scenarios had multiple sources, including point sources, and the area/volume sources may not have been dominant (Personal communication with Catheryn Price, CERC, 1 February 2006). CERC also refer to validation of the line and volume sources in ADMS-Urban and ADMS-Roads, but these tests were exclusively for road traffic emissions, not industrial releases. This raises concerns about the applicability of area and volume source algorithms in ADMS3 for industrial odour releases.

AERMOD and ADMS treat area sources in different ways: AERMOD assumes a virtual upstream point source, whilst ADMS describes dispersion from elements within the source. Significant differences in results are to be expected. Field validation for area sources in isolation have not been found.

Previous discussions with Chris Sidle (Environment Agency) concluded that stacks exceeding 75 m were very rare for installations of interest to the Environment Agency with respect of odour impact **. Area sources of odour are predominantly less than 5 m in height.

** Modelling scenario scoping previously carried out for earlier specification for Project 3.

Some 'new-generation' dispersion models cannot include building downwash effects at all for area sources. If buildings are present near area sources and relevant receptors are also within close range, there may be no wisdom in attempting modelling at all if the models are being used to the extremes of their capabilities in such scenarios.

Therefore, the following building and source configurations are ranked *in order of decreasing certainty* with respect to dispersion modelling predictions:

- Elevated stack with no nearby buildings;
- Low stacks/near ground level point sources with no buildings;
- Elevated stacks with buildings;
- Low stacks/near ground level point sources with buildings/ Ground level area sources with no buildings;
- Ground level or low area sources in the presence of buildings.

On this basis, it must be recognised that the modelling of low stacks or area sources, where nearby buildings are present, is subject to large uncertainties. A risk-based approach is needed: if the predicted odour concentration is low, then even with model uncertainty there might not be a problem.

Establishing such a hierarchy for treating model uncertainty is not a new concept. R. Timmis (Personal communication in Fisher 2001) has proposed a hierarchy for treating model uncertainty in dispersion calculations. In straightforward cases of a flat terrain and no buildings, one could assume that models are accurate within a factor of 2 (or some other value accepted by the modelling community). For the more complex cases, additional factors of uncertainty arise (perhaps further factors of 2) to account for the additional uncertainty (in the ways models account for buildings, for example). There is a concern that this could soon lead to very large factors of uncertainty as one multiplies the factors together; in practice however, additional complexity usually just involves a single extra process. Sometimes complexity is not necessarily multiplicative and bounds on possible ranges arise.

If after allowing for this hierarchy of uncertainty, the range of results from two different models still do not overlap, the difference between the models is deemed to be *significant*. The reasons for a *significant difference* are varied. The models may be using different algorithms, e.g. plume behaviour near the top of the boundary layer, or near the top of a hill.

Fisher (2001) recognised that the treatment of uncertainty is inevitably associated with some subjective judgements. Using results from more than one model has distinct advantages in decision making, if used with care.

5.11 Peak-to-mean ratios

It is important to note that the Draft H4 Indicative Odour Exposure Standards do not include any dependence on peak-to-mean ratios. The Environment Agency's view is that statistical relationships between modelled percentiles of *hourly average* concentrations and surveys of the level of annoyance currently provide a more robust method of determining odour exposure (Phase III).

Fluctuations are the differences between a specific dispersion occurrence (or "realisation") and the ensemble mean of all such occurrences as given by a dispersion model. The relationship between the ensemble mean and fluctuating plume concentrations has been described in many papers (Hall and Kukadia 1994, CERC 1998). Conventional dispersion models struggle to treat short-term variability; this is discussed by Best *et al.* (2004), where various methods for accounting for fluctuations in a crude way (e.g. peak-to-mean ratios) are summarised.

Peak-to-mean ratios do not account for the fundamental differences between the distribution of the mean and fluctuating components of concentration in a plume. An important feature of the distribution of concentration with time at a particular point in a dispersion plume is that it does not have a nominal occasionally recurring 'peak' value, only the *probability* that some particular value will occur.

While more work is required into the relationship between odour nuisance and the detection of odour over short time-scales, it has been proposed that it is unlikely that a highly intermittent detectable odour with a *mean* concentration well below the detection threshold would cause a nuisance (Simms *et al.* 1999). Consequently, the effects of very short-term fluctuations (over a matter of seconds) can probably be adequately modelled by considering an averaging period of 3 minutes or similar. Some regulatory authorities (e.g. in Australia) have set odour thresholds based on a 3-minute averaging period for modelling.

Draft H4 recognises that there are a number of areas in odour modelling which need further validation, such as peak-to-mean ratios (Appendix 4 Part 1) and appropriate averaging times. It has been shown that there are potentially large differences between area near-ground level sources and tall point sources in terms of the downwind fluctuation to which a receptor would be exposed. Elevated point sources can produce considerably greater peak-to-mean ratios than area sources.

There is other evidence that the frequency and magnitude of peaks in odour concentration over timescales of seconds and minutes are determining factors for the perception of odour (Phase III). At low concentrations, a rapidly fluctuating odour is often more noticeable than a steady background. Small scale meteorological variations mean that the peak values can be some ten times higher than the 1-hour mean values for point sources within the same general weather situation.

The Draft H4 guidance is at variance with the guidance in New Zealand (Ministry for the Environment 2002) which advises that an 'annoyance threshold' (typically $5 \text{ ou}_{\text{EM}}^{-3}$ for the mean concentration) is divided by a peak-

to-mean ratio. It is noteworthy that the New Zealand ‘annoyance threshold’ is a ‘Type 1’ standard derived from first principles, not through community dose-response surveys. The peak-to-mean ratios are different for area sources, stacks without building wake effects and stacks with building wake effects. The peak-to-mean ratios are different in the near field and the far field. A summary of theoretical peak-to-mean ratios for the various circumstances is presented in Literature Review for Project 2 (Environment Agency 2007b), and is reproduced in Table 5.10.

Table 5.10 Summary of Published Peak-to-Mean Ratios set out in Literature Review for Project 2

Type of source	Near-field	Far-field
Area source	2 – 2.2	2.2 – 2.6
Tall stack, no buildings	0.14 - 0.3	0.8 – 1.6
Stack with building downwash	2.2	2.2
Ground level point source *	0.2 -0.4	0.7 – 1.7

In the UK, a peak-to-mean ratio of 10 has in the past been recommended (Woodfield and Hall 1994) for certain models (notably the D1 stack height calculation screening method) which calculated hourly mean concentrations, with the flexibility that it could be reduced where dispersion was significantly affected by buildings, or for relatively low stacks. As explained in the Literature Review for Project 2, it is not currently the intention of Project 3 to recommend the use of peak-to-mean ratios in dispersion modelling of odours.

Peak-to-mean ratios in relation to their application to assessing odour near building wakes are discussed by Schauburger and Piringer (2004). However, this study views the use of peak-to-mean ratios as a screening tool only, and not as an alternative to more sophisticated modelling and assessment techniques. In a modelling study for a sewage treatment works, the hourly mean prediction of ISC3 were converted to ensemble mean 3-minute averages using a power law (Witherspoon *et al.* 2000).

5.12 Fluctuations Models

There are 2 different approaches to setting odour benchmarks, which are described in the Literature Review for Project 2:

- Type 1: Theoretical odour modelling guidelines attempt to incorporate from first principles the FIDOL factors (Phase III). Since the perception of odour occurs over time-scales of less than an hour (a few seconds), such theoretical odour modelling guidance would need to be based on modelling that takes fluctuations into account.

* This is an unusual case for odorous emissions

- Type 2: Empirical odour modelling standards are derived from the empirical relationship between odour exposure and annoyance which has actually been measured by a community survey. The Indicative Odour Exposure Standards in Draft H4 are derived from this method. The fluctuations in odour concentrations are accounted for in the *community response* in this case, rather than up-front in the dispersion modelling or exposure predictions.

A number of numerical models have been developed for predicting the fluctuation concentrations in dispersing plumes. Fluctuation models attempt to calculate specific occurrences (or “realisations”) of concentrations around the ensemble mean value. Most of these models predict the bulk statistical properties of the plume, so are of more limited assistance in dealing with questions of intermittency and the frequency of repetition of fluctuating peaks. Neglecting the effect of turbulent fluctuations in modelling can lead to the underestimation of these peak values and hence of the overall short-term impact of a release.

It is important to note that in contrast with ensemble mean odour concentrations, the frequencies of high concentration events due to fluctuations do not scale up or down proportional to the odour emission rate. Thus a halving of emissions will not halve the frequency of high concentration events above a given concentration value due to fluctuations.

Conventional models predict the average concentration over a period of an hour or more. Normal atmospheric turbulence is sufficient to cause local wind velocity to vary rapidly in both speed and direction and thereby to cause rapid fluctuations in the concentrations of an airborne pollutant at a fixed receptor position. It is difficult to predict the short-term peak concentrations with any confidence, but it is possible to derive a probability distribution function (PDF) of concentration and to determine what fraction of this PDF exceeds the benchmark or threshold (Lee and Stewart 1999). This approach is attempted by models such as Omonos^{*} and the fluctuations facility in ADMS3 (Dyster *et al.* 1999).

Omonos can model single and multiple, point and area sources. Like ADMS3, this model predicts the likelihood of the concentration exceeding the threshold value. Omonos makes use of the meandering plume model approach. Omonos uses the Pasquill-Gifford curves (or their mathematical equivalents) to generate the hourly mean concentrations; it takes this output then calculates fluctuation intensities and intermittency (J.R. Stewart, Queen’s University of Belfast, personal communication, January/February 2006).

When modelling multiple point sources, the total 1-hour mean concentration at any receptor is additive. However, probabilities of exceeding the odour benchmark cannot be obtained by summing the concentration from individual sources. Furthermore, addition does not address the situation where *instantaneous* concentrations for each point source lie below the threshold concentrations but their sum exceeds the threshold level. Omonos includes a numerical method for combining probabilities from multiple point sources which

^{*} Developed by QUESTOR at Queens University of Belfast

incorporates the concentration fluctuation behaviour of each individual plume. A modified method allows for the prediction of probability of exceeding the benchmark from an area source.

The basis of odour assessment in Germany is the 'odour hour' (Janicke *et al.* 2004). By definition, an hour is an 'odour hour' if, within this hour, odour is perceived for at least 10% of the time. Odour assessment is then based on the frequency of occurrence of 'odour hours' within a year.

Concentration fluctuation models are also being tested in other countries for odour regulation (Lofstrom 2004). The Lagrangian dispersion model AUSTAL2000G has been developed for calculating the frequency of these concentration fluctuations over an hour. The odour dispersion model of the Finnish Meteorological Institute (ODO-FMI) has the ability to represent the short-term peak values of odour concentration. It uses the meandering plume method for point sources and probability density function modelling for area and volume sources (Pietarila and Savunen 2004).

Comparison of 'odour hour' modelling results against odour frequency measurements have been presented for a piggery (Janicke *et al.* 2004), using the German regulatory AUSTAL2000G Lagrangian dispersion model. The field validation exercise released a passive tracer (SF₆) simultaneously with the odorous emission, to measure the ensemble mean hourly concentrations, in order to relate the fluctuating concentration to the ensemble mean, and derive a 'perception function'. The same scenario was also investigated using wind tunnel modelling. Consideration should be given to carrying out investigations of this nature in UK-specific situations, with the commonly used 'new-generation' models.

There is a difficulty in that detailed modelling (e.g. using fluctuations) is likely to be inappropriate for many odour sources due to the wide potential variation in odour emission rates and hence difficulty in obtaining accurate source input data. It would be inappropriate to aim for ever-increasing sophistication in modelling outputs when the reliability and confidence in the input data (particularly that pertaining to the odour emission rates, Phase I) is low. Moreover, buildings cannot be accounted for in ADMS3 when modelling fluctuations. Concentration fluctuations will be significantly modified when material becomes incorporated within a building wake region. The developers of ADMS (CERC, Simms *et al.* 1999) state that the additional mixing involved will tend to reduce the peak concentrations, giving a more uniform, less intermittent structure to the concentrations. Entrainment in the turbulent mixing cavity behind a building is particularly common at wastewater treatment works, and has been the subject of discussion.

Also, there are few supporting social surveys for the dose predicted using fluctuation modelling. On the other hand, it may be argued that the response to odour in the social/annoyance surveys includes the effect of concentration fluctuations. A person's perception will inevitably incorporate his/her experience of *fluctuations* rather than any experience of ensemble means. This is because ensemble means are a form of statistical summary and so do not generally occur in any particular time and place in the field.

Validation of any models against predictions of odour concentration is almost impossible, because it is not possible to measure ambient odour concentrations in real time using DDO. Limited attempts have been made to measure fluctuations in concentrations using SO₂ or other tracers over time-scales of less than an hour.

5.13 High percentiles

For the 98th percentile prediction, the model requires an entire year of hourly emission rates. Appendix 4 of Draft H4 Part 1 states that when the odour emission rate and/or other model input parameters fluctuate, then an assessment of this variation should be undertaken.

Although the Draft H4 Indicative Odour Exposure Standards are set as the 98th percentile of hourly means, many studies have highlighted the significance of the 100th, 99.9th and 99th percentiles. The perception and annoyance associated with an odour (of particular character) may in fact be related to these percentiles, as well as the 98th percentile. The relationship between these high percentiles and the 98th percentile needs to be investigated. The relationship depends on stack height and model choice, and needs to be investigated further in dose-effect studies in the UK. The level of annoyance in the community may be dictated by the percentiles higher than the 98th percentile; it may be that complaints are actually driven by the top 0.5 % of hourly mean concentration. This is discussed in the Literature Review for Project 2.

The uncertainty of a model increases for higher percentiles (Tables 5.5 and 5.9). Report P4-095/TR (Environment Agency 2002b) states that in the UK, the 99th percentile is roughly twice the 98th percentile, but the source of this finding is not transparent.

Quoting model outputs as percentiles implies a far greater confidence in model predictions than is reasonable. It may be more useful to report the number of occurrences/prediction of hourly concentration above a benchmark in any year of meteorological data.

Different values of percentile compliance are in use around the world. Miedema (1992) indicated that higher percentile concentrations were best correlated to odour annoyance when the emission source was active for less than 50% of the time. Therefore, for highly variable and intermittent sources, the 99.9th percentile concentration may be a stronger determinant of odour annoyance than the 99.5th percentile.

The Literature Review for Project 2 (Environment Agency 2007b) suggests that there is little convincing evidence to support the use of any particular percentile component. The percentage exceedance calculated by the model does not necessarily mean that odour nuisance will occur for all those hours, and reasons are set out in the Review.

German field work (Both and Koch 2004) suggests that *frequency* is the overwhelming factor in determining annoyance, more important than odour intensity; how often you are annoyed is more important for complaints than how

acutely you are annoyed at the time when annoyance occurs. This is because frequent annoyance is more likely to detract from people's quality of life/amenity than occasional acute annoyance.

However, most odour modelling guidelines take both frequency and intensity (usually via concentration) into account, for example an hourly odour concentration below the relevant Draft H4 Indicative Odour Exposure Standard, to be predicted for at least 98% of the hours in the year.

A recent, published inter-comparison of 'new-generation' models by AQMAU (Sidle *et al.* 2004) included ADMS3.1, AERMOD and AERMOD PRIME. This study concluded that there was greater agreement between the models for the 98th percentile than higher percentiles. For this reason, and in the absence of any UK specific dose-response studies investigating higher percentiles for annoyance correlation, **it is recommended that, for the present, Draft H4 maintains the 98th percentile as the parameter for odour concentration prediction.**

Dutch case studies reported by Miedema (1992) indicated that higher percentile concentrations were best correlated to odour annoyance when the emission source is active for less than 50% of the time. Therefore, for highly variable and intermittent sources, the 99.9th percentile concentration may be a stronger determinant of odour annoyance than the 99.5th. Dose-response studies in England and Wales are awaited before Draft H4 can consider the use of other percentiles for odour assessment.

A new assessment parameter has been proposed in Denmark that recognises that short and tall stacks complying with the same maximum concentration limit as a percentile will probably result in different odour exposure. This is because the critical meteorological parameters occur more often for low stacks than for tall stacks. It has been proposed that a single unifying assessment parameter is used that accumulates all odour concentrations above the odour thresholds and weights the frequency of the individual odour concentrations with the intensity (which is proportional to the log concentration).

5.14 Meteorology

Dispersion behaviour depends critically on the state of the atmosphere. However, the availability of meteorological data is limited. The atmosphere is subject to natural variability, within a given, defined state or on a statistical basis from one year to the next (Hall *et al.* 1999). The way in which meteorological data is pre-processed prior to the model carrying out the dispersion calculations can have a significant effect on the predictions of the model.

It is likely that differences between models reflect the uncertainties in estimating turbulent dispersion in the atmospheric boundary layer. Low wind speed and very stable conditions are difficult to represent in any dispersion model, and in these situations, the predicted peak ground level concentrations can vary widely. The difficulties which commonly-use models have with dealing with low wind speed/stable conditions are well documented, and therefore the uncertainty can be taken into account, albeit qualitatively. It follows that this can

give rise to predictions of high percentile concentrations that also vary widely between models.

Draft H4 recognises that the meteorological data obtained from a Met. Office observing station and used for dispersion modelling may not always reflect very localised conditions for example a valley location producing its own inversions. These conditions would not be picked up even by 5 or more years of meteorological data from a remote observing station; Environment Agency guidelines on modelling for PPC purposes stipulates the use of 5 years of meteorological data. Where unusual weather conditions are suspected as being the cause of odour occurrences, it may be appropriate to model specific hours of theoretical weather conditions, rather than to rely solely on the long-term meteorological observation datasets. However, it would be necessary to know how often these conditions occurred, and what emissions they coincided with in time.

A local meteorological station on site could be of value if time and resources allow for its operation over at least a year. It could identify the very unusual/extreme events which are causing the odour episodes. However, it would need to record the necessary parameters, including some parameters from which atmospheric stability could be estimated.

A discussion of difference in dispersion calculations due to using two sources of meteorological data can be found in Hall and Spanton (1999). One source recorded the wind speed for an hour as the average over the first 10-minutes of the hour; the other source actually averages over the entire hour. A 1-minute averaged wind speed can be expected to show a greater variability than an hourly average, which could affect the calculation of the 98th percentile concentration. However, the study concluded that year-to-year variability in wind speed measurements showed greater differences in predicted concentration than those which arose from the different ways in which wind speeds were derived for each hour.

The uncertainties in meteorological pre-processing for dispersion models has been investigated (Middleton and Thomson 2002). AERMOD and ADMS treat day/night transition in different ways, the models put different limits on how stable or unstable conditions can get. Urban effects can reduce the frequency and severity of stable conditions at night, and different models may treat this in different ways. The complexities of turbulent flows over real surfaces are not fully described by any dispersion model.

A sensitivity analysis of model input parameters for multiple odour sources at a wastewater treatment works showed that year of meteorology had little effect on the predictions, surface roughness being more significant. This study found that assumptions on source strength, and the lack of error margins for the emission estimates (i.e. the empirical equations in the Sewage Treatment Odour Prediction (STOP) model), gave most cause for concern. The author concluded that it was essential to confirm theoretical estimates by checking conditions on site.

Hall *et al* (2000b) identified that the different meteorological pre-processors in ADMS and AERMOD were part of the reason for very different predictions of pollutant concentrations under certain scenarios. A detailed investigation into the uncertainty in model predictions due to meteorological data input and the subsequent treatment of that input within Gaussian models was commissioned by ADMLC in 2003 (Auld *et al* 2003). Roughness length was a parameter whose contribution to overall model uncertainty was significant. It was found that peak concentrations often occur during periods of low wind speeds (particularly high concentrations occurring when wind speeds are less than 1 ms^{-1} , i.e. calms), and agreement between models (ADMS and AERMOD) under these conditions is poor. Poor agreement is also found during dusk periods and periods where there is rapid change in cloud cover. If the model user is particularly concerned with modelling these conditions, or where results are close to benchmark concentrations, they are recommended to consider obtaining predictions from more than one model. It was recommended that the use of Numerical Weather Prediction (NWP) meteorological data should be considered when the nearest surface observing station may be unrepresentative of the source location.

ADMLC has also funded research of the options for the most appropriate meteorological data for use in short-range dispersion models (ADMLC 2001). However, the NWP data did tend to produce higher maxima in predicted concentrations compared to the model outputs when using conventional observed meteorological data. It was concluded that the use of NWP data for modelling studies, subject to validation in the future, could be a good substitute for observed data when the latter was not available. This option is likely to become more useful in the future as the NWP data improves further and observing stations continue to close down or become automated.

Stuetz and Frechen (2001) have noted that many models have a minimum applicable wind speed, typically 1 ms^{-1} . For many odour sources (e.g. waste water treatment works) the highest odour concentrations are likely to occur for stable conditions when wind speeds are low and dispersion is poor. Hence the higher concentrations are likely when the wind speed is less than 1 ms^{-1} and outside the range of model applicability. Hourly average wind speeds of less than 1 ms^{-1} may be expected to occur for more than 2% of the time at many locations, causing concern for the prediction of the 98th percentile odour concentrations. It is necessary to examine how “calms” are treated (or ignored) by the met pre-processing and dispersion model being employed.

Environment Agency guidance on dispersion modelling for PPC states that at least 5 years of meteorological data should be used for modelling any particular installation. Inter-year variability has been shown to be in the range of $\pm 30\%$ for the annual mean, and in a similar range for the high percentiles. This enables the uncertainty due to meteorology to be ranked relative to other component uncertainties (see Figure 5.2).

5.15 Terrain height

Most research and development of dispersion modelling in the presence of terrain features has been conducted by the US EPA, and is well documented. Validation exercises for the terrain modules in AERMOD are continuing.

The commonly expressed rule-of-thumb is that terrain with slopes below about 10% has only limited effects on dispersion. On this basis, most of the UK is topographically flat and so terrain is of limited interest in most UK dispersion studies. Previous discussion with Chris Sidle (Environment Agency) identified that in terms of odour emitting process, building downwash effects were more prevalent than complex terrain effects. However, in the minority of areas where terrain is more severe its effect on dispersion can be very marked and it can then be the critical feature in the determination of an adequate stack height or abatement.

The problems of low emission heights within hilly terrain have been highlighted by Lohmeyer *et al.* (2004a). Valley drainage flow can result in poor dilution due to very little vertical dispersion. Dispersion models are poorly validated for this type of meteorology. Though rare in the UK, cases such as these for sources with significant potential for odour annoyance may merit wind tunnel modelling (which however can usually only replicate neutral conditions) or CFD (Lohmeyer *et al.* 2004b).

The local weather data available from the Met Office (or other organisation) for the purposes of dispersion modelling does not always reflect very localised conditions e.g. a valley location may produce its own inversions. Where this type of local effect is leading to increased exposure (i.e. increased frequency of a recognisable odour, or increased annoyance due to a more intense odour), it may be appropriate to model short-term weather conditions individually, ideally those identified by means of an on-site weather station.

Detailed model inter-comparisons of ADMS, AERMOD and ISC were carried out by Hall *et al.* (2000a) including numerous terrain profiles and stack locations. These have been reviewed in detail with respect to their relevance for odour impact assessment. ISC showed considerably higher predicted high percentile concentrations compared to the other models for most terrain profiles and stack siting. The deficiencies of ISC for terrain calculations are recognised, which has led to the development of AERMOD.

There are large, non-systematic differences in the predictions between new-generation models when significant terrain features are included in the dispersion calculations. These models are not well validated for this type of modelling, the predictions are associated with high levels of uncertainty, and they can be highly sensitive to small changes in how the terrain is defined in conjunction with other model set-up parameters. A risk-based approach is needed if modelling terrain; if the predicted odour concentration is low, then even with model uncertainty there might not be a problem.

5.16 Location of receptors

Draft H4 recognises that there may be long-term plans for off-site receptors (e.g. housing) closer to the installation than exist at present. The guidance states that the operator should have due regard to the possible temporary nature of the situation and ensure that improvement is factored into longer-term plans for the locality near the site, e.g. potential land-use changes in the Local Plan. This gives some steer to modelling odour concentrations at the point of maximum impact, in addition to modelling concentrations at specific receptors representing existing housing, site boundary etc.

Moreover, dispersion models are better at predicting the maximum concentration than where it occurs in space; the point of maximum impact and the odour prediction at this point may be inspected for its proximity to existing sensitive locations. Currently, Draft H4 emphasises the need to model odour impact at 'sensitive receptors'.

Sometimes, complaints can be received from an individual, or small group, when those around them seem to be unaffected. This could be due to different sensitivities or other local or highly subjective issues.

5.17 Summary of the uncertainties in the modelled exposure

The aim of this study is not so much to get a precise estimate of the overall uncertainty, but rather to investigate the component uncertainties. It should be possible to identify, for example, if the dispersion modelling has a much more significant component uncertainty than emissions quantification, or assignment to different bands of unpleasantness. Chapter 5 of this research report has described the uncertainties associated with the dispersion modelling, which is an input to the next stage, correlation with annoyance. It is not necessary to quantify in detail all the component uncertainties. By convention, they can be considered insignificant if less than 10% of the largest uncertainty.

There are some component uncertainties that can be quantified. Sensitivity analyses can be carried out readily; for example, studies on the effects of meteorology (choice of year, site) are well documented. Other component uncertainties are very case-dependent, and no general uncertainty estimate can be provided (e.g. the effect of buildings or complex terrain).

There are, therefore, a number of key choices that a practitioner needs to make at this phase of the Draft H4 assessment methodology. These choices focus on the components with the greatest uncertainty:

- Is dispersion modelling an appropriate tool for the source/s under consideration?
- How many, and which, dispersion models to use?
- How reliable is the information on the temporal variability in source strength?

- What level of detail is required for the sensitivity analysis on the odour source strength?
- Is the model fit for purpose, e.g. if complex effects are present?

These choices will have a fundamental effect on the uncertainty of the modelled exposure, and will contribute to the overall uncertainty of the Draft H4 assessment.

The main focus of this study is on controlled point-source releases of odour, because this is the main application of the Draft H4 modelling approach to back-calculate permissible ELVs. Figure 5.2 shows a summary of the best estimates of component uncertainties in modelling exposure for a very simple, hypothetical scenario. It uses as an example the Draft H4 98th percentile of hourly means as the model output. The potential errors and uncertainty in many of the components of dispersion modelling have been investigated and documented, such that numerical estimates are available and are therefore included in several of the boxes in Figure 5.2.

It is clear, at least in qualitative terms from the discussion in Chapter 4, that the uncertainty on the source strength value used as an input to the dispersion model will dominate the uncertainty on the predicted 98th percentile odour concentration, in most cases. In the modelling exercise itself, the choice of model is the most significant uncertainty, though this can be reduced relatively easily by the use of more than one dispersion model. In terms of ranking, choice of meteorological observing site, choice of meteorological year/s, and model inputs such as source velocity and temperature, are of lesser significance as long as appropriate care/expert opinion is exercised when making these choices.

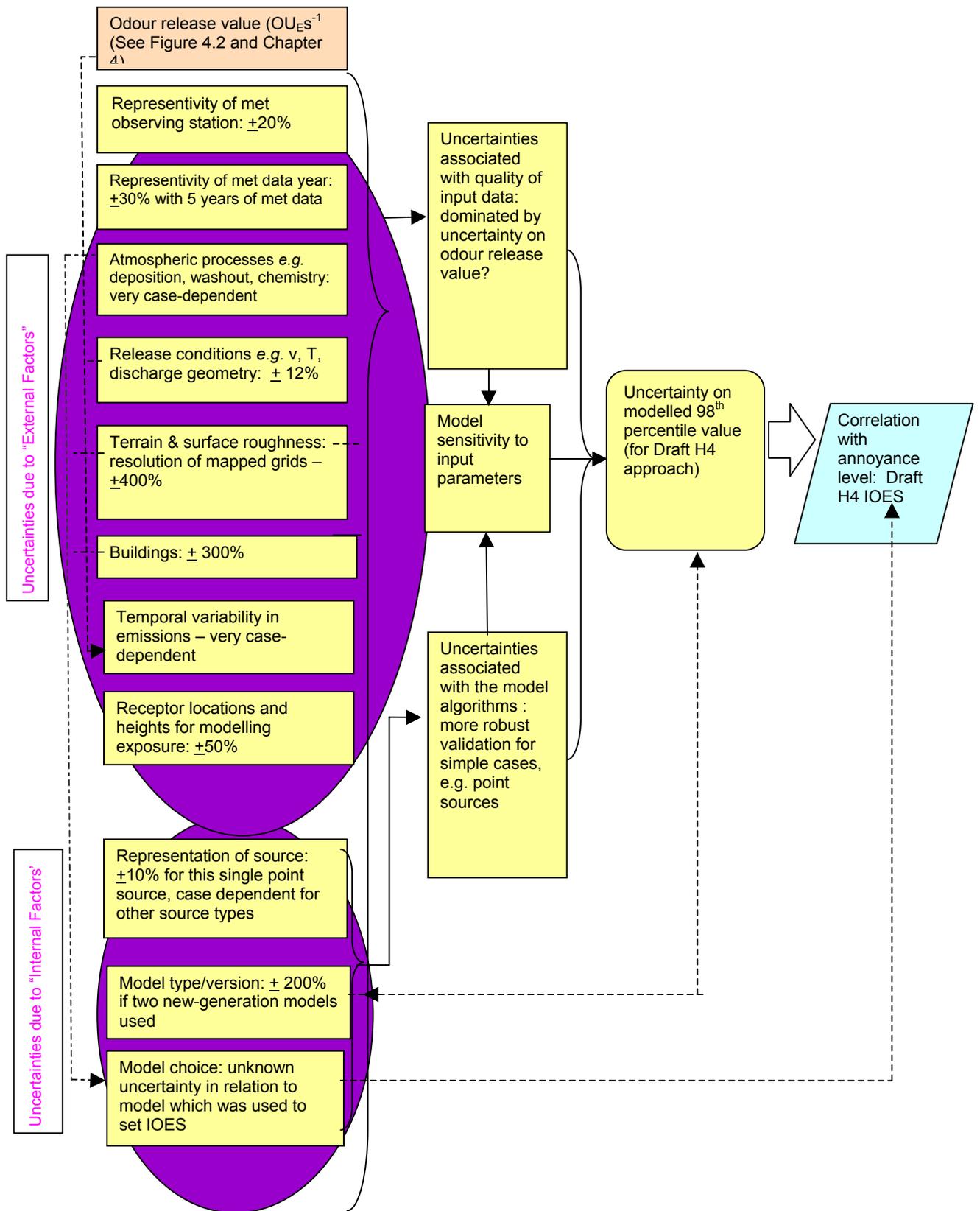


Figure 5. 2 Summary of the estimated uncertainties in the modelled exposure value: hypothetical simple scenario

6 Phase III: Correlation with Annoyance

6.1 Overview of this phase

The PPC Regulations include in their definition of pollution “*emissions as a result of human activity which...cause offence to any human senses*” and the Environment Agency has given special consideration as to how the endpoint of odour “offence” may be anticipated, measured and assessed in terms of annoyance.

The preceding phases – quantifying an odour release value and then using this as input to an atmospheric dispersion model – provide us with a prediction of the odour exposure at the receptor location. This exposure is given in terms of the 98 percentile of 1-hour average concentrations ($C_{98, 1\text{-hour}}$)[#].

However, knowing the exposure, on its own, is not sufficient. From this predicted **odour exposure**, a view must be formed on whether it is likely to cause **odour annoyance**[◇]. This can be done by establishing what is, in effect, a calibration curve: the basis for establishing this correlation is the empirical relationship between odour exposure (measured, or more usually, modelled) and annoyance (measured by a community survey). The details are given in Project 2 of this cluster.

The approach used by the Environment Agency in Draft H4 when determining Applications/Variations under PPC is to derive from this empirical epidemiological relationship a numerical modelling guideline standard to define in numerical terms its “benchmark” criterion of “*no reasonable cause for annoyance*”. The odour exposure predicted by dispersion modelling is compared to the numerical modelling guideline standard to judge whether annoyance is likely to occur. The aim of odour control is therefore to ensure there is “*no reasonable cause for annoyance*”.

[#] This encompasses (in a different form) the frequency, intensity, duration and location aspects of the FIDOL factors described earlier in Chapter 3; the offensiveness (i.e. unpleasantness) component is based on a consensus judgement for that particular type of installation or odour.

[◇] This difference between odour exposure and odour annoyance is crucial. Similarly, it is important to emphasise the technical differences between annoyance and nuisance. This chapter concerns itself only with odour exposure and annoyance.

^{*} For the purposes of the PPC Regulations, the Agency deems the point at which pollution in the form of offence to the sense of smell is occurring to be the point at which there is “*reasonable cause for annoyance*”. The “benchmark” criterion of “*no reasonable cause for annoyance*” does not necessarily equate to no complaints - it is designed to be a level of exposure that a high proportion of the exposed population, with normal sense of smell, finds “acceptable” on a long-term basis. Conversely, the lack of complaint should not necessarily imply the absence of an odour problem, as there will be an underlying level of annoyance before complaints are made

Draft H4 allows two routes to making this comparison with its benchmark:

- The first route is to derive a bespoke odour exposure guideline standard from an industry-specific dose-response relationship carried out in England and Wales.
- The alternative is to use the default “Indicative Odour Exposure Standards” that have been derived by the Environment Agency from the dose-response relationship established for a community living around a livestock installation in the Netherlands. The assumption has been made that the results of this study can be applied generically to other applications with certain adjustments and factors applied (see Section 6.3 for more details).

The Indicative Odour Exposure Standards approach clearly has an additional layer of uncertainty compared to the approach of deriving a bespoke guideline standard. For this reason the two approaches are looked at separately in the following sections.

6.2 Bespoke odour exposure guideline standards derived from industry-specific dose-response studies

6.2.1 Summary of the component uncertainties in industry-specific guidelines

The Environment Agency’s Draft H4 guidance allows* PPC applicants to derive industry-specific dose-response relationships between annoyance and 98% concentrations (1-hour average), as an alternative to using the indicative exposure standards provided (which are effectively “default values”). At the time of writing, the Environment Agency had not received any applications in England and Wales that used bespoke industry-specific dose-response relationships.

In order to investigate the uncertainty in the correlation of odour exposure with annoyance based on such a dose-response study, it is necessary to examine the component steps in the study. These can be summarised as follows:

- i. A suitable field-study candidate is identified. The installation should be representative of the industry or odour type in question, and the surrounding community should be representative of the wider population to which it is going to be applied (e.g. the whole sector).
- ii. Odour emissions from the installation are measured at source by dynamic dilution olfactometry. Then, atmospheric dispersion modelling is

* It is perhaps worth noting that in the New Zealand guidance (Ministry for the Environment New Zealand 2003) that post-dates the draft H4, a stronger steer is given: industry is expected to derive their own dose-response relationships and it is made clear that the indicative guideline values provided there are temporary and only for use until such studies have been completed.

- carried out to predict the surrounding ground-level concentrations ($C_{98, 1\text{-hour}}$).
- iii. Zones of distinct odour exposure levels are identified and the percentage of people annoyed in each zone is estimated using questionnaires and a random sample of addresses falling in each zone.
 - iv. The percentage of “annoyed” respondents in the sample is then plotted against the exposure band to establish a dose-effect relationship for the particular type of installation/odour. The dose-response curve, which tends to be S-shaped, is fitted by regression (see Figure 3.2 for an example).
 - v. A suitable level of annoyance – which can be chosen based on policy grounds or technical grounds – is selected and the corresponding level of odour exposure is read-off from the curve. This 98% concentrations (1-hour average) is the bespoke odour exposure guideline standard.

Each of these steps contains component uncertainties that will go to make up the uncertainty in the bespoke odour exposure guideline standard that is finally derived. These are summarised in Figure 6.1, and the information available on these component uncertainties is discussed in the following sections.

6.2.2 Uncertainties associated with the application of the dose-response study

Temporal uncertainties

The main temporal uncertainty is where the period of the dose-response study is not wholly representative of the period of interest. For example, it is known that most odour complaints are received during the summer months; this can be because high temperatures favour odour formation at source, but it is also because people spend more time in their gardens and with their windows open. The selection of an appropriate duration for the study depends on many of the same factors used to select the duration of a representative ambient air quality monitoring campaign; these are discussed in Environment Agency Technical Guidance Note M8 (Environment Agency 2006). Careful planning is needed to be ensure the study period encompasses (or even focuses on) those plant conditions and meteorological conditions that are expected to lead to odour episodes.

This uncertainty component will be specific to the particular dose-response study being carried out; it is difficult to put any kind of general figure on its uncertainty. In practice, the approach taken is likely to be one that ensures this component uncertainty is minimised – by ensuring that such studies adhere to robust Environment Agency technical guidance – and then considered negligible.

Spatial uncertainties

Due to the effort, complexity and cost of a bespoke dose-response study, it is not likely that these will be carried out for every installation. Certainly the expectation of regulators in Australia and New Zealand is that such studies will require central funding or will be carried out by operators on a sector-wide basis. It is important therefore that the installation chosen for the study, and the odours it releases, are representative of that sector as a whole. Uncertainty will be greater where the sector encompasses many different types of plant and odour; it will be smaller where the sector focuses on a narrow band of processes and odours with similar characteristics. Regarding differences in odour unpleasantness between some types of installation, Project 2 in this cluster considers this issue in more detail, discussing whether the grouping of some industries/sectors needs to be broken down further to account for this (Environment Agency 2007b).

A further contributor to the spatial uncertainty will be how representative are the community surrounding the installation chosen for the dose-response study. Project 2 in this cluster discusses the differences in annoyance response* between people who are used to a type of odour and those who are not; and also those people who have some kind of economic relationship with the installation – for example, they are employed there. In addition, other ambient stressors, such as noise, dust and crowding can influence the response (Environment Agency 2007b, see Figure 3.1). These factors can lead to significant differences between the annoyance response to a given odour exposure at the study location and the responses at other installations in that sector at different geographical locations. This is emphasised in New Zealand guidance (Ministry for the Environment 2002, 2003) which recognises that in community response-based studies an odour-modelling guideline is determined for a particular site based on population annoyance data, and therefore is based on the tolerance of an existing community to an existing industrial or trade activity. The tolerance of an *existing* community to a *new* industrial or trade activity, or increased odour emissions from an existing activity, would be expected to be lower. The New Zealand guidance therefore applies a tolerance factor to reduce (i.e. make more stringent) the odour-modelling guideline to represent the lower tolerance of existing communities to new sources of odour. Based on case studies, the tolerance factor was estimated at approximately 2–5.

Again, it is not possible to place any numerical estimates of the magnitude of these situation-specific uncertainties, but practitioners must make themselves aware of the possible importance and form a qualitative view on the significance of this component of uncertainty. Robust Environment Agency technical

* The factors affecting human response to odour, and the relationship between perception, cognitive appraisal and behavioural responses, are also summarised in Appendix 2 of Draft H4 Part 1, and described in detail in Environment Agency R&D Report P4-095/TR (Environment Agency 2002b). Draft H4 recognises that the setting of odour benchmarks is still a developing field. The response to exposure to an odour is primarily subjective – how strong it is, what it smells like, and in what context. There can be large differences between individuals in terms of their sensitivity, likes and dislikes and attitudes to exposure.

guidance on carrying out sector-wide dose-response studies will help minimise this component uncertainty.

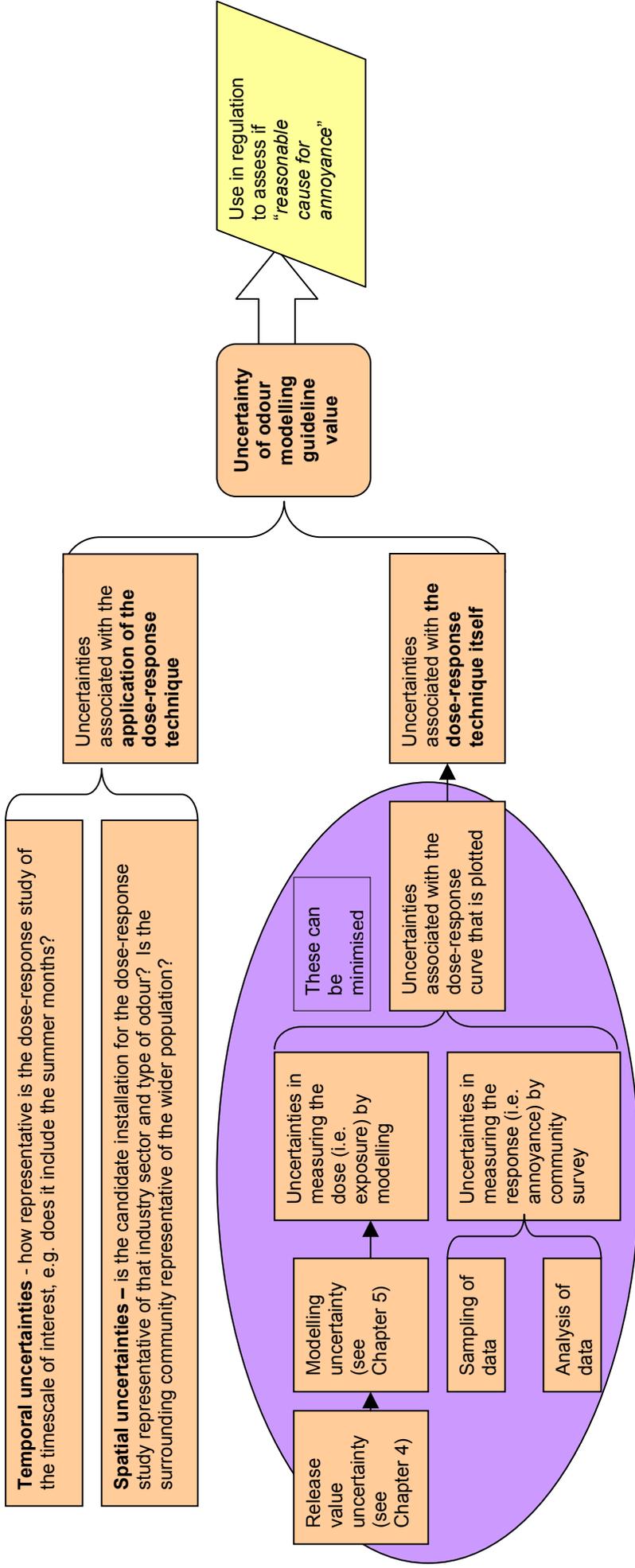


Figure 6.1 Summary of the component uncertainties in assessing odour annoyance from a bespoke dose-response study

6.2.3 Uncertainties associated with the dose-response study technique itself

The odour modelling guideline is read off from a dose-response curve, essentially a calibration curve, established by plotting two parameters:

- the odour exposure – the predicted ground-level concentrations ($C_{98, 1\text{-hour}}$) in the surrounding areas; and
- the odour response – the percentage of people annoyed in each zone

The odour exposure values are obtained by atmospheric dispersion modelling of the source odour release value for the installation chosen for the study. All the release value and modelling uncertainties described in Chapters 4 and 5 will therefore contribute.

The odour response values are obtained from a community survey. Zones of distinct odour exposure levels are identified around the installation and the percentage of people annoyed in each zone is obtained from questionnaires and a random sample of addresses falling in each zone. A well established VDI methodology (VDI 1997) exists for this.

No studies in England or Wales have yet been carried out for PPC regulatory purposes, so information on the likely uncertainties to be expected have to be drawn from studies carried out elsewhere. For the dataset from a study around a Dutch piggery*, Van Harreveld (2004) describes how regression fitting an S-shaped curve to these data showed a strong correlation ($r > 0.9$) between modelled exposure and annoyance. A level of 10% annoyed was chosen as the lowest level that would be statistically significant, based on the “background noise” for measurement of annoyance using questionnaires plus two times the standard deviation. The level of 10% annoyance to pig odours correlated with an exposure ($C_{98, 1\text{-hour}}$) of $1.3 \text{ ou}_E \text{ m}^{-3}$ and this was used for the basis for setting an odour modelling guideline of $1.5 \text{ ou}_E \text{ m}^{-3}$.

The dose response curve is, essentially, a calibration curve for estimating a value of X from Y. The uncertainty on this estimate can be obtained by calculating the “inverse confidence limits”, which takes account of the uncertainty in the position of the calibration line itself and also the fact that the measured value of Y is subject to some uncertainty. The standard deviation of the estimate of X from Y, for example the odour exposure value corresponding to 10% annoyance, is given by an equation. This is described in a Royal Society of Chemistry Technical Brief (2006), which also gives macro programs for carrying out the calculation.

The original data from the Dutch study described above were not available for this study. It was not possible (or within the scope of this literature review) to re-analyse the original data and carry out statistical analysis, so no estimate is available for the uncertainty in the derived modelling guideline. It is recommended that the Environment Agency should strengthen the technical guidance given in Draft H4 to require this uncertainty to be calculated for any

* This study was used for the data underpinning the Draft H4 Indicative Odour Exposure Standards (see Section 6.3) and also the Irish EPA odour exposure criteria for livestock odours.

bespoke odour modelling guidelines produced from sector-specific dose-response studies.

6.3 Using default Indicative Odour Exposure Standards

6.3.1 Background to the Summary Indicative Odour Exposure Standards

The Environment Agency recognises the reality of the situation as it currently exists – no bespoke dose-response studies have been carried out in England and Wales for PPC regulatory purposes and there is likely to be a time lag between issuing strong guidance on this being the favoured route, and such studies being completed to provide sector-specific odour exposure modelling standards. Also, such community dose-response studies do require a definable odour source and an existing community with sufficient population density to represent a suitable case study. This may not always be possible.

To fill this gap the Environment Agency developed, for Draft H4, default “Indicative Odour Exposure Standards” that were derived from the relationship established between ground-level odour concentration and odour annoyance for a sample of test subjects living around a livestock installation in the Netherlands. These “default” exposure criteria are set as a 98-percentile, 1-hour average concentration ($C_{98, 1\text{-hour}}$) of 1.5, 3.0 or 6.0 $\text{ou}_E \text{m}^{-3}$ for high, medium and low categories of odour unpleasantness, respectively.

The assumption was made that the results of this study could be applied generically to other applications with certain adjustments and factors applied. For this reason, the use of Indicative Odour Exposure Standards to assess if there is “*reasonable cause* for annoyance” has an additional layer of uncertainty compared to the approach of deriving a bespoke guideline standard. The background to developing the Indicative Odour Exposure Standards is described below along with a discussion of these additional uncertainties.

The derivation of the Indicative Odour Exposure Standards is described in detail in the earlier Environment Agency Report P4-095 *Assessment of Community Response to Odorous Emissions* (Environment Agency 2002b), which formed the main background work for Draft H4. As discussed earlier, the dataset that formed the main underpinning for the proposed values was collected in the Netherlands for livestock odours (Bongers *et al.* 2001) as specific data for England and Wales were not available[#]. (This data set was also used by the Irish EPA as a starting point to derive odour exposure criteria for livestock odours.) In brief, odour emissions from a piggery were measured at source by

[#] The Environment Agency research noted that, ideally, the dose-effect relationship for UK citizens in UK conditions should be assessed experimentally to confirm the findings obtained abroad, but as of date this has not been carried out.

dynamic dilution olfactometry, then modelled to predict the surrounding ground-level concentrations ($C_{98, 1\text{-hour}}$). Zones of distinct odour exposure levels were identified and the percentage of people annoyed in each zone was estimated using questionnaires and a random sample of addresses falling in each zone. The percentage of “annoyed” respondents in the sample was then plotted against the exposure band to establish a dose-effect relationship for livestock odours. Regression fitting of an S-shaped curve showed a strong correlation ($r > 0.9$) between modelled exposure and annoyance. A level of 10% annoyed was chosen as the lowest level that would be statistically significant, based on the “background noise” for measurement of annoyance using questionnaires plus two times the standard deviation.

For the general public, the level of 10% annoyance to pig odours correlated with an exposure ($C_{98, 1\text{-hour}}$) of $1.3 \text{ ou}_E \text{ m}^{-3}$ and this was used for the basis of the most stringent Draft H4 indicative criteria, for high offensiveness (i.e. unpleasant) odours, of $1.5 \text{ ou}_E \text{ m}^{-3}$.

The earlier Environment Agency research suggested that it would be preferable to use the measurement of annoyance potential to characterise odour emissions, rather than using $\text{ou}_E \text{ s}^{-1}$ alone, for input to the dispersion modelling and comparison with the percentage annoyed respondents to establish a dose-effect relationship. This would allow the true effect of hedonic tone, unpleasantness and odour character to be included in the relationship for different types of odour. However, a laboratory method for measuring annoyance potential had not then been developed, so it was proposed that existing rank-order data for industrial odours (as shown in Table A1.1 of Draft H4) should be the basis for assigning different odour types into a simple three-band categorisation:

- High odour annoyance potential (e.g. animal rendering, fat and grease processing, etc.)
- Medium odour annoyance potential – all odours not in categories High or Low.
- Low odour annoyance potential (e.g. bakeries, coffee roaster)

The particular numerical guidelines that were assigned in Draft H4 to the indicative criteria for odours of medium unpleasantness and odours of low unpleasantness were arrived at as follows.

For residents in areas where pig odours were a common feature, the 10% annoyed level corresponded to an exposure of ($C_{98, 1\text{-hour}}$) of $3.2 \text{ ou}_E \text{ m}^{-3}$ and this value was used for the basis of the Draft H4 indicative criteria for mildly unpleasant odours of $3.0 \text{ ou}_E \text{ m}^{-3}$. The most lenient Draft H4 indicative criterion of $6 \text{ ou}_E \text{ m}^{-3}$, assigned to “less offensive” odours, was based on 10% annoyed of respondents who worked in agriculture (corresponding to $13 \text{ ou}_E \text{ m}^{-3}$) combined with data from a dozen dose-effect studies for industrial sectors in the Netherlands where the 10% annoyed level corresponded with approximately $< 5 \text{ ou}_E \text{ m}^{-3}$. In addition, inspection of a number of consultancy projects indicated that between 90 and 95% of complaints registered for wastewater treatment and solid waste management occurred in the exposure range of 5 to $10 \text{ ou}_E \text{ m}^{-3}$.

A summary of how the Indicative Odour Exposure Standards were derived from the Dutch results is shown in Table 6.1.

Table 6.1 How the Indicative Odour Exposure Standards relate to the Dutch study results

Results of Dutch livestock dose response study		Indicative Odour Exposure Criteria for Draft H4 inferred from these results.
1.3 ou _E m ⁻³ was equivalent to 10% annoyance of general public to pig odours	→	1.5 ou _E m ⁻³ chosen as limit for industry sectors with odours considered “more offensive”
3.2 ou _E m ⁻³ was equivalent to 10% annoyance of residents to pig odours in areas where pig odours were a common feature	→	3 ou _E m ⁻³ chosen as limit for industry sectors with odours considered “mildly offensive”
13 ou _E m ⁻³ was equivalent to 10% annoyance to pig odours of respondents who worked in agriculture, combined with data from a dozen dose-effect studies for industrial sectors in the Netherlands	→	6 ou _E m ⁻³ chosen as limit for industry sectors with odours considered “less offensive”

6.3.2 Comments on the additional uncertainties with Indicative Odour Exposure Standards

As is obvious from the summary of the Indicative Odour Exposure Standards in Table 6.1, it was necessary to make a number of assumptions on the applicability of the research data to conditions in England and Wales and to industries other than intensive livestock. Using a calibration curve for livestock odours and applying it to other odours and installations will give an additional layer of uncertainty compared to deriving a modelling guideline from a bespoke dose-response study.

There is a question over the modelling carried out on the original Dutch study, and how comparable the results are to ‘new-generation’ models used by practitioners today. This is discussed in Chapter 5. For example, it is not known what local conditions were assumed for the modelling, which would have affected the uncertainty, e.g. buildings, terrain, types of source, variability of emissions. If the dose-effect relationship was established mainly for livestock, there is concern about its transferability to industrial-type odours.

Apart from this, the additional uncertainties fall into the category of spatial uncertainties described earlier for bespoke studies (see Section 6.2.2). The main sources of this spatial uncertainty are:

- how representative (or indeed correct) is the assignment of industry-sectors and odour types to the three bands of unpleasantness;
- differences in the odour intensity:concentration relationship between different odours; and
- any relative differences in community responses and odour tolerances.

Assignment of industry-sectors and odour types to the three bands of unpleasantness

The proper assignment of different industrial odour mixtures to one of the three bands or categories of unpleasantness was one of the main drivers for Project 2 in this cluster and is discussed in detail in that report.

The Environment Agency's Draft H4 Indicative Odour Exposure Standard ($C_{98, 1\text{-hour}}$) of $1.5 \text{ ou}_E \text{ m}^{-3}$ for "unpleasant" odours was based on an actual dose-response study for livestock (pig odours). However, the Indicative Odour Exposure Standards of $3.0 \text{ ou}_E \text{ m}^{-3}$ for "mildly unpleasant" odours and $6.0 \text{ ou}_E \text{ m}^{-3}$ for "least unpleasant" odours are not so robust. These were *not* derived from bespoke dose-response studies of industrial odours of different unpleasantness. Rather, the concentration values chosen were based on dose-response curves for receptors of differing sensitivities to the same livestock odours, as was summarised in Table 6.1. This raises the questions of:

- a) Whether the apparently empirical selection of concentration factors to give the exposure benchmarks of ($C_{98, 1\text{-hour}}$) $3.0 \text{ ou}_E \text{ m}^{-3}$ for "mildly unpleasant" and $6.0 \text{ ou}_E \text{ m}^{-3}$ for "least unpleasant" odours are appropriate. (Though this question is outside the scope of this study/review.)
- b) Exactly how the relative unpleasantness of livestock odours should be categorised: in the research work the dose-response curve for livestock (pig odours) was used to define the exposure benchmark of $1.5 \text{ ou}_E \text{ m}^{-3}$ for "unpleasant" odours; in contrast, Table A6.1 in Draft H4 categorises intensive livestock rearing odours as falling in the medium category of "mildly unpleasant".

Project 2 of this cluster summarises some possible options to account for the unpleasantness or hedonic tone of a single source (or multiple sources of the same hedonic tone – evaluation of multiple sources of different hedonic tone would be more complex. Such approaches are however no substitute for a representative dose-response study.

Differences in the odour intensity: concentration relationship between different odours

The intensity of an odour as experienced by a human receptor is related to the log of the odour concentration. This means that if the concentration is increased ten-fold, the perceived intensity will increase by a smaller amount. The odour concentration-intensity (OCI) relationship is specific to the particular odorant. The practical effect of this is that at concentrations above the odour threshold (i.e. above $1 \text{ ou}_E \text{ m}^{-3}$) the perceived intensity – which affects annoyance response - is different for different odours. Some species such as hydrogen sulphide cause a rapid increase in perceived intensity. Other odours, such as toilet air fresheners and perfumes, are designed to be perceived at a similar intensity regardless of dilution, so they give rise to only a slow rise of perceived intensity with increasing concentration. This is discussed in detail in Project 2 of this cluster (Environment Agency 2007b).

This is illustrated in Figure 6.2, which shows the relationship between the perceived intensity and the odour concentration for two compounds, hydrogen sulphide and butanol (Department of Environmental Protection 2002). Hydrogen sulphide has a higher specific intensity than butanol and so is perceived as a stronger odour at the same concentration. So if an odour concentration of 10 odour units was chosen as the appropriate modelling guideline, then butanol would be perceived as a weak odour, whereas hydrogen sulphide would be perceived as a distinct odour. To have equivalent protection against different odours would require choosing an *intensity* level for the numerical odour guideline and then working across the graph to determine the appropriate concentration for that odorant. Using Figure 6.2 as an example, if the guideline was set at a “distinct” perceived odour (in the laboratory) then the appropriate concentrations would be 11 and 33 odour units for hydrogen sulphide and butanol respectively (Environment Agency 2002c).

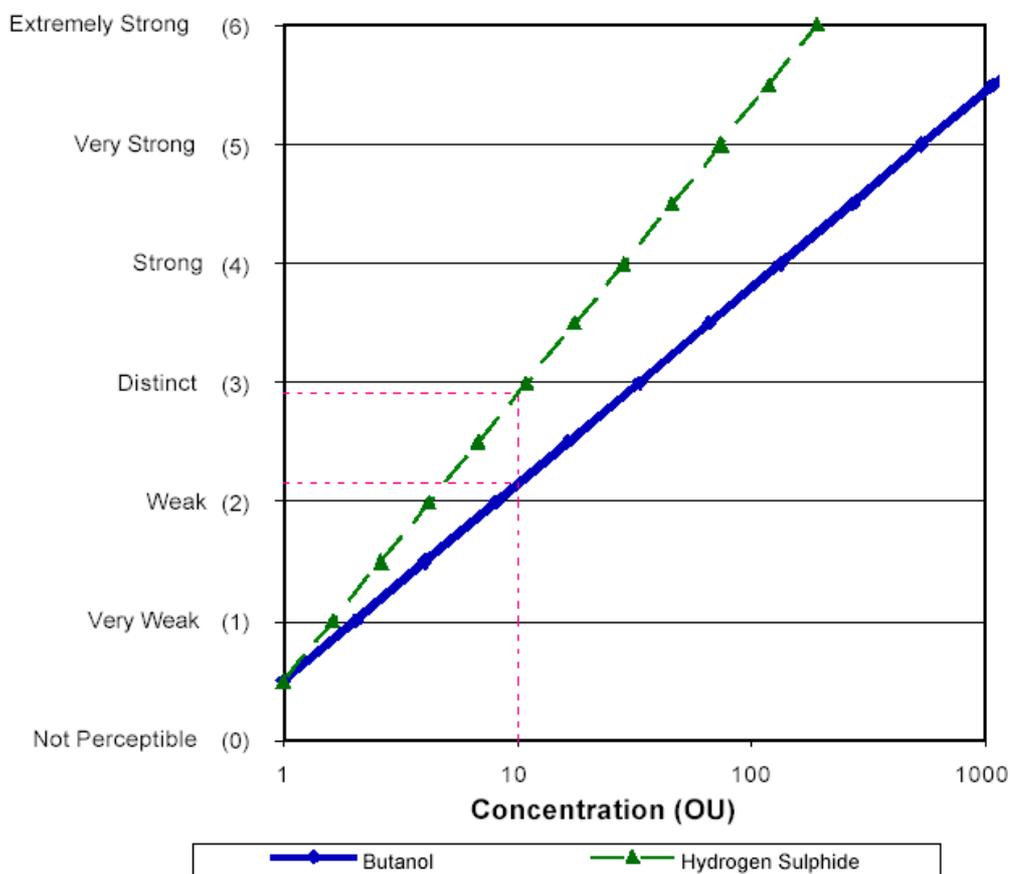


Figure 6.2. Relationship between perceived odour intensity and odour concentration for butanol and hydrogen sulphide

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Where a bespoke dose-response study is carried out, the relationship between concentration and perceived intensity is accounted for in the community annoyance response, but where the calibration curve derived from one odour is applied to different odours, or multiple sources, this could have an impact on uncertainty. This uncertainty is specific to the situation (i.e. the differences in the

OCI relationships for the two odours in question) so it is difficult to give a generally applicable value for this (and for this reason it has been argued that it would be better to use the odour intensity value instead of concentration in any numerical benchmark, as is the practice in Australia). But in the above example there is a 300% difference between the concentrations required to give a “distinct” perception of odour intensity.

Relative differences in community responses and odour tolerances.

In using a calibration curve for livestock odours and applying it to other odours and installations, there are likely to be differences in the communities surrounding the installations. For example, communities exposed to livestock odours are in rural areas, whereas other installations may be in urban areas where communities could have different tolerances. Another source of uncertainty derives from using community response data from one country (the Netherlands) and assuming the same response would occur in England and Wales.

6.4 Summary of uncertainty in correlation with annoyance

The uncertainties in the correlation of odour exposure with odour annoyance will be a combination of:

- *Random component uncertainties* – these are precision-type uncertainties, responsible for the scatter and the correlation coefficient found for the dose-response curve; and
- *Systematic component uncertainties* – these are the bias or accuracy-type uncertainties that include factors such as how relevant the Dutch pig-odour response curve is to other odour types in English/Welsh conditions, and how appropriately the concentration levels for the “offensiveness” bands have been set.

If a series of dose-response studies had been carried out under conditions prevalent in England and Wales, it would have allowed the repeatability of the Draft H4 method to be estimated. Unfortunately no such studies have been carried out. Instead, this review has used the uncertainty-budget approach to look at the component uncertainties that are likely to make up this phases of the assessment.

Figure 6.3, showing the component uncertainties from bespoke dose-response studies, has been annotated with as much information as is currently available, but many gaps will remain until the data from bespoke studies in England and Wales are available.

Regarding the use of Indicative Odour Exposure Standards, it is clear from the preceding discussion that using a calibration curve derived from Dutch livestock odours and applying it to other odours and other types of installation in different countries will give an additional layer of uncertainty compared to deriving a modelling guideline from a bespoke dose-response study. However, again it is

not possible to place any numerical estimates of the magnitude of this additional layer of uncertainty, as it is situation-specific. For a livestock installation the additional layer of uncertainty might be expected to be small. For other, very different, odours or installations the leap of faith is wider and the additional uncertainty may be much larger. Practitioners must make themselves aware of the possible importance and form a qualitative view on the significance of this component of uncertainty.

On the positive side, the level of annoyance measured by the survey in the New Zealand Technical Report (Ministry for the Environment 2002) was found to be consistent with the odour dose-community-response curves reported by Miedema (1992). The dose-response curves, although developed for other industries and using a Dutch community response, appeared to be valid for pulp mill odours in New Zealand.

In terms of the future outlook for using Indicative Odour Exposure Standards compared to deriving a modelling guideline from a bespoke dose-response study, the Draft H4 guidance does state that the above benchmarks are indicative standards and that dose-effect studies in England and Wales are planned. It also states elsewhere in the document that *“the only realistic way of estimating the actual level of annoyance in a particular community resulting from exposure is by carrying out dose-response studies locally”*. However, Draft H4 appears much less explicit than the New Zealand guidance in highlighting the “interim” nature of these generic-type odour guidelines and that they should ideally be superseded by industry-specific guidelines developed from bespoke dose-response studies. It is possible that some dose-response studies will be performed around waste management facilities as part of a study into defining loss of amenity through odour carried out as part of Defra’s Waste Research R&D programme*. There is also a possibility of UK Water Industry Research (UKWIR) coordinating some studies around wastewater treatment plants to support the water industry in meeting the Defra *Code of Practice on Odour Nuisance from Sewage Treatment Works* (Defra 2006).

* Details at <http://www.defra.gov.uk/environment/waste/wip/research/index.htm>

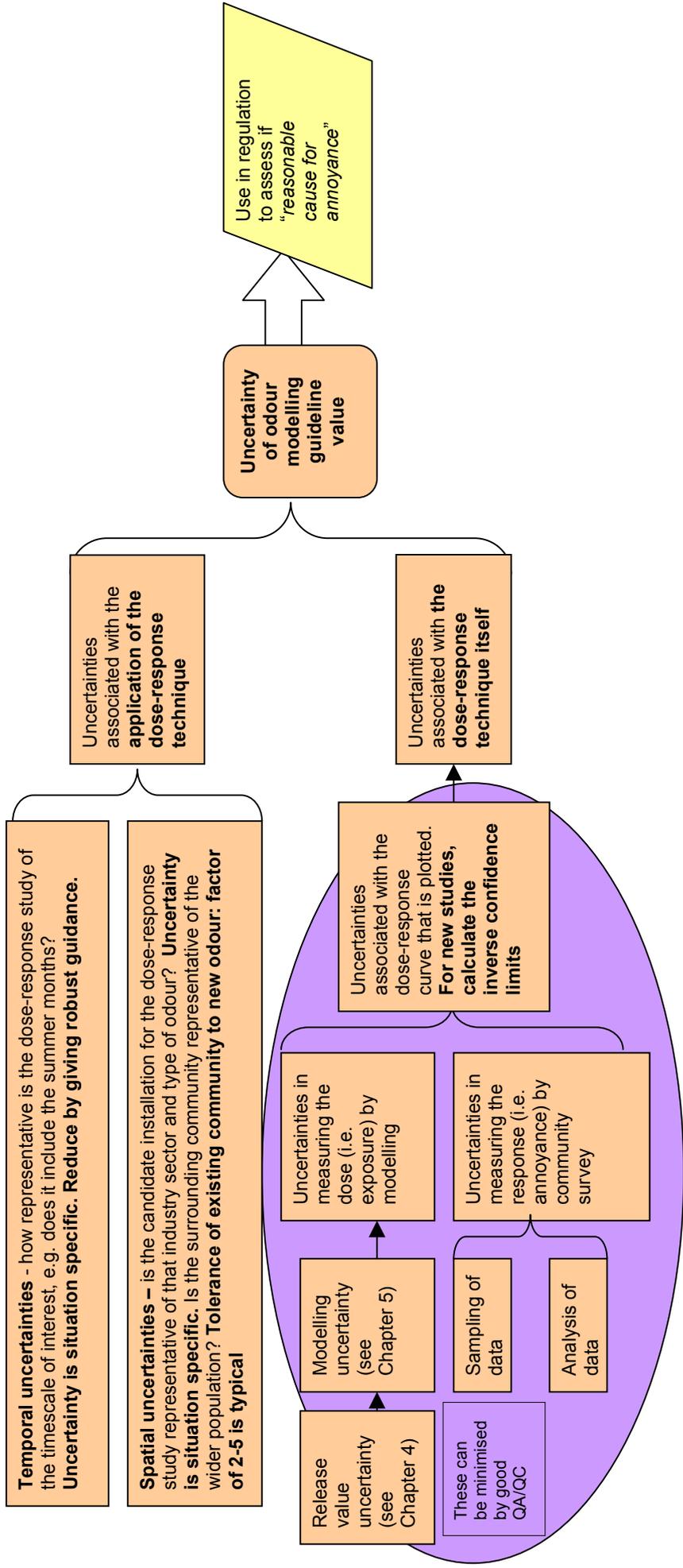


Figure 6.3 Summary of best estimates of component uncertainties in assessing odour annoyance from a bespoke dose-response study

7 Phase IV: – Correlation with Negative Coping Behaviours (Nuisance and Complaints)

7.1 Adverse outcomes of odour exposure – annoyance, nuisance, complaints

The terms used to describe an odour's potential to cause these adverse effects, and the terms describing the effects themselves (e.g. annoyance and nuisance), are explained in detail in Project 2 of this cluster. A summary of the technical differences between annoyance and nuisance is shown in Table 7.1, which is drawn from that project. It is helpful to read these terms in conjunction with Figure 3.1, which summarises the pathway from odour formation through to complaint.

It is important to emphasise the technical differences between annoyance and nuisance: one major distinction is that annoyance occurs from an immediate exposure; nuisance on the other hand, is caused cumulatively, by repeated events of annoyance.

Both, or either, annoyance and nuisance can lead to complaint action. However a lack of complaints does not necessarily prove there is no annoyance or nuisance. On the other hand, there needs to be an underlying level of annoyance before complaints are generated.

A complicating factor is that as well as the technical definition, nuisance is also a term in law (e.g. Statutory Nuisance). The definition of Statutory Nuisance covers seven areas, and that which relates to odour is (s.79(1) Environmental Protection Act (EPA) 1990):

“any dust, steam, smell or other effluvia arising on industrial, trade or business premises and being prejudicial to health or a nuisance;”

The legal use of nuisance has preceded the technical definition of nuisance described here, which has only relatively recently been put forward and generally accepted. The EPA 1990 contains no technical definitions of nuisance, such as maximum concentrations, frequencies or durations of odour in air.

Any technical guidance can currently, therefore, only provide a method to assess nuisance against currently accepted technical criteria. Although it is hoped that these would also satisfy the legal definition of Statutory Nuisance for odour, it is ultimately only the Court that can make that decision.

Although the extension of the Draft H4 methodology to predict nuisance or complaint is beyond the scope of this literature review, a brief overview has been given.

Table 7.1. Proposed technical definition of annoyance and annoyance potential (Van Harreveld 2001)

<i>Annoyance potential</i>	<p>Annoyance potential is the attribute of a specific odour (or mixture of odorants) to cause a negative appraisal in humans that requires coping behaviour when perceived as an ambient odour in the living environment. It is an attribute of an odour that can cause annoyance or nuisance.</p> <p>Annoyance potential indicates the magnitude of the ability of a specific odorant (mixture), relative to other odorants (mixtures), to cause annoyance in humans when repeatedly exposed in the living environment to odours classified as ‘weak’ to ‘distinct odour’ on the scale of perceived intensity (VDI 3882:1997, part 1).</p> <p>Whether annoyance potential of an odour does, or does not, cause annoyance (see below) depends on location and receptor factors.</p>
<i>Annoyance</i>	<p>Annoyance is the complex of human reactions that occurs as a result of an immediate exposure to an ambient stressor (odour) that, once perceived, causes negative cognitive appraisal that requires a degree of coping.</p> <p>Annoyance may, or may not, lead to nuisance and to complaint action.</p>
<i>Nuisance</i>	<p>Nuisance is the cumulative effect on humans, caused by repeated events of annoyance over an extended period of time that leads to modified or altered behaviour. This behaviour can be active (e.g. registering complaints, closing windows, keeping “odour diaries”, avoiding use of the garden) or passive (only made visible by different behaviour in test situations, e.g. responding to questionnaires or different responses in interviews). Odour nuisance can have a detrimental effect on our sense of well-being, and hence a negative effect on health. Nuisance occurs when people are affected by an odour they can perceive in their living environment (home, work-environment, recreation environment) and:</p> <ul style="list-style-type: none"> i) the appraisal of the odour is negative; ii) the perception occurs repeatedly; iii) it is difficult to avoid perception of the odour; and iv) the odour is considered a negative effect on their well-being.
<i>Nuisance potential</i>	<p>Nuisance potential is the characteristic of an exposure situation, which describes the magnitude of the nuisance that can be expected in a human population when exposed to an odour intermittently, but over an extended period of time, in their living environment. Nuisance potential is a function of many factors, such as the attributes of the odorant (mixture) in question, the frequency and dynamics of variation of the exposure (caused both at source and as a result of atmospheric dispersion) and attributes of the specific population that is exposed.</p>
<i>Nuisance sensitivity</i>	<p>Nuisance sensitivity is an attribute of a specific population (or an individual) that indicates the propensity, relative to that of other individuals or populations, to experience nuisance when exposed to an odour intermittently, but over an extended period of time, in their living environment.</p>

Note: these definitions are from a technical perspective to enable a scientific understanding of the odour impact process. They are not legal or regulatory definitions (e.g. “no reasonable cause for annoyance”, or Statutory Nuisance).

7.2 Correlation of exposure with nuisance and complaints

Chapter 6 gave a detailed review of the uncertainties involved in the correlation of predicted odour exposure with annoyance. In theory, there is no reason why a dose-response study could not be carried out using a different response measurand – either nuisance or complaints. For nuisance, this would need to be preceded by work to establish a method for measuring this parameter.

Dose response studies using complaints as the response measurand have been carried out in New Zealand and Australia, as described in Project 2 of this cluster (Environment Agency 2007b). For example, the case study of a sewage treatment facility in Sydney reviewed in the New Zealand Technical Report (Ministry for the Environment 2002) compared modelling results to areas of varying levels of complaint (as have most other similar investigations in Australasia). The study established the link between observed levels of complaint and percentile odour concentrations. The odour concentrations of the plant emissions were also measured using European methods, and ambient concentrations were modelled using AUSPLUME and also using a wind tunnel. The community was alerted to the study by means of regular advertising, and so the community response may have been enhanced. Nevertheless, the study found that for sewage plant emissions, 99.5 percentile concentrations below 46 ou m^{-3} were unlikely to lead to complaint. The equivalent certainty threshold concentration for this study was 5 ou m^{-3} . The application of a 2 to 5 tolerance factor to convert this to a guideline for assessing a proposed new activity indicates a modelling guideline for new sewage plants near sensitive areas in the range of 1 to 3 ou m^{-3} (1-hour average, 99.5%, certainty).

Environment Agency report P4-079/TR/2 (2002c) concluded that for stack discharges there are insufficient complaints data to validate the use of dispersion models in the UK. The report states that to achieve validation, there is a need to record the number of complaints from a single stack for approximately 2 years. The report does however state that VDI 3883 Parts 1 and 2 (questionnaire and community survey techniques for odour annoyance assessment) have been recently employed to assess odour levels in the vicinity of a large industrial complex in Bedfordshire. The findings of this study could prove useful to the review of uncertainties for Project 3.

Draft H4 recognises that complaint records are useful for looking at the distribution of complaints around a source over a period of time. Identifying locations on a local map can show the footprint of a problem. Bureau Veritas carried out an odour dispersion modelling study in 2002 for a chipboard manufacturing plant with a single stack. The ADMS3.1 model was used to plot contours of the 99th and 98th percentile odour concentrations, where a peak-to-mean ratio of 10 was used to convert 1-hour mean model outputs to short-term fluctuations. It was found that the 98th percentile contours for $5 \text{ ou}_E \text{ m}^{-3}$ and above encompassed 16 complainants out of 27 households. It was found that the 99th percentile contours for $5 \text{ ou}_E \text{ m}^{-3}$ and above encompassed 17 complainants out of 27 households. It is noteworthy that the result of ADMS3.1 fluctuation modelling did not give such a good agreement against the complaints records, with the model predicting a fluctuation concentration

exceeding $5 \text{ ou}_E \text{ m}^{-3}$ during at least 25 of the hours in the year at only 3 of the 27 complaining properties. This arose from the inability of ADMS3.1 to model building downwash effects in conjunction with the concentration fluctuations.

A dispersion modelling study using AERMOD PRIME 4.4 for a refinery carried out in 2006 predicted percentiles ranging from 90th to 100th, with an assessment benchmark of $1.5 \text{ ou}_E \text{ m}^{-3}$. It was found that the complaints records (which spanned 3 years) matched the 99.9th and 100th percentile contours best. This is consistent with the assumption that annoyance is represented by the 98th percentile, and actual complaint action would be prompted by a higher frequency of the odour at the benchmark concentration; there is an underlying level of annoyance before complaints are made i.e. individuals experience annoyance before they express this annoyance in the form of a complaint.

7.3 Complaints analysis

In contrast to the predictive approaches for assessing odour impact (e.g. dispersion modelling), direct approaches include complaints monitoring and community assessment.

There are a number of factors which affect how and when complaints are made, and this makes it difficult to use complaints as an accurate reflection of the overall level of annoyance in a community. There tends to be a 'threshold' which has to be reached before initial complaints are made, but thereafter complaints may be made more readily. The number of people actually experiencing the effects of annoyance caused by odour appears to be much higher than the number of registered complaints. In the Netherlands, the annoyance caused by odour is investigated by systematic year-on-year surveys (which disguise the purpose to avoid bias). Consideration should be given to establishing surveys of this kind in the UK.

Comprehensive reviews of incidence of complaints in relation to odour exposure for the UK are not available. However, in the UK, an increasing number of complaints of odour have been received by many industries, and the general consensus of opinion is that it will continue to increase (Environment Agency 2002c).

Complaints analysis is not covered by any standard method or recognised protocol. Complaints registration provides an insight into the prevalence of a *symptom* of odour annoyance, not in the prevalence of the annoyance itself. There are many factors at play that determine the ease or difficulty of registering a complaint.

7.4 Summary of uncertainty in correlation with nuisance and complaints

Although the extension of the Draft H4 methodology to predict nuisance or complaint is beyond the scope of this literature review, a brief overview has been given.

The legal use of nuisance has preceded the technical definition of nuisance described here, which has only relatively recently been put forward and generally accepted. The EPA 1990 contains no technical definitions of nuisance, such as maximum concentrations, frequencies or durations of odour in air. This would need to be addressed, together with a nuisance measurement methodology, before any estimate of the uncertainty in the correlation with odour exposure could be made.

Complaints are more usually measured directly by complaints monitoring, rather than predicted. However, dose response studies using complaints as the response measurand have been carried out in New Zealand and Australia, but using different models to those in common use in the UK and using different percentiles to describe exposure.

It would be safe to assume that the uncertainties in correlating predicted exposure with either nuisance or complaints levels would be considerably higher than for annoyance, due to the additional factors involved.

8 Main Findings

Draft H4 sets out an approach for quantifying odour emissions, dispersion modelling to estimate the odour exposure, and correlation of the predicted exposure against the expected degree of annoyance (using Indicative Odour Exposure Standards). There are uncertainties associated with each step of this assessment process. This study has examined the component uncertainties by means of a detailed literature review, and identified their relative significance. An attempt is made at ranking the component uncertainties where possible.

There is a need to emphasise that the Draft H4 monitoring-modelling approach is suitable only for “normal” odour emissions, and not for occurrences of abatement failure, out of control conditions, etc., where huge temporal variations in odour emissions rates might be expected. Moreover, it is most unlikely that monitoring would be carried out under these conditions. Therefore, the analysis of uncertainty here does not consider such scenarios.

No analysis of uncertainty can take account of the Draft H4 monitoring-modelling approach being mis-applied or applied badly. For instance, an uncertainty budget cannot take into account practitioners making measurements at the wrong (i.e. not representative) parts of the process cycle, any more than it can take account of errors in transcribing numbers from instruments to notepad, or the mixing-up of samples for analysis. Rather, such failures should be prevented or minimised by working to a relevant QA/QC regime. For the estimation of the source emission rate, the obvious route for this would be to ensure all such odour measurements for regulatory purposes are carried out under the MCERTS scheme.

8.1 Source strength

The dispersion model requires the odour source strength as a key input. The most reliable value for this input could theoretically be determined from a large number of periodic, dynamic dilution olfactometric (DDO) measurements, on a single existing chimney, where the release is controlled, continuous, and does not vary with time or process cycle. The uncertainty escalates sharply for estimated odour emission rates (e.g. for a proposed process), time-varying emissions, multiple sources on a site, and particularly when specific compounds are used as surrogates for the total odour.

DDO is not currently practical on a continuous basis for any source. The inability to accurately quantify the odour emission temporal variation, and difficulties in correlating the source variation with time-varying meteorology in the dispersion modelling, is a significant source of uncertainty in many assessments.

There are many components in the derivation of the source strength value for which a numerical estimate of uncertainty cannot be quoted. These are very situation-specific. The uncertainty on the source strength value can be several orders of magnitude even for commonly-encountered situations with time-

varying emissions and/or estimates based on surrogate compounds or emission factors.

8.2 Dispersion Modelling of Exposure

There is a need to recognise that the uncertainties associated with modelling some types of odorous release (e.g. diffuse/fugitive area sources, non-vertical vents) are very large. In such cases, the use of dispersion modelling as an assessment tool may not be an appropriate use of resources.

The results of different, new-generation models can vary greatly, even for simple cases. Dispersion modelling is usually carried out when the risk of odour annoyance is high. Under these circumstances, the use of more than one dispersion model is justified for a risk-based approach.

Almost no details are available on the dispersion model which was used to establish the Dutch empirical dose-response curve, nor the input data for that modelling. Of particular concern is the reliability of the source strength data which were used for the Dutch livestock modelling.

There are many components of the dispersion modelling for which a numerical estimate of typical uncertainty cannot be quoted. These are situation-specific, e.g. when complex terrain is present.

Dispersion models are currently in practical use only for predicting ensemble mean (typically hourly mean) concentrations. Fluctuation modelling is not yet adequately validated. As long as this remains the case, the 'Type 2' approach for odour assessment set out in Draft H4 (hourly mean modelling compared against an empirical benchmark) must remain the only feasible option.

ADMFC has provided detailed guidance on best practice for dispersion modelling. Most of the guidelines are applicable to odour modelling. However, currently there is neither consistency nor uniformity in the way that model sensitivity and uncertainty are expressed in permit applications to the Environment Agency.

8.3 Correlation with annoyance

The uncertainties in the correlation of odour exposure with odour annoyance will be a combination of:

- *Random component uncertainties* – these are precision-type uncertainties, responsible for the scatter and the correlation coefficient found for the dose-response curve; and
- *Systematic component uncertainties* – these are the bias or accuracy-type uncertainties that include factors such as how relevant the Dutch pig-odour response curve is to other odour types in UK conditions, and how appropriate the concentration levels for the “offensiveness” bands have been set.

If a series of dose-response studies had been carried out under UK conditions, it would have allowed the repeatability of the Draft H4 method to be estimated. Unfortunately no such studies have been carried out. Instead, this review has used the uncertainty-budget approach to look at the component uncertainties that are likely to make up this phases of the assessment.

Regarding the use of Indicative Odour Exposure Standards, it is clear from the preceding discussion that using a calibration curve derived from Dutch livestock odours and applying it to other odours and other types of installation in different countries will give an additional layer of uncertainty compared to deriving a modelling guideline from a bespoke dose-response study. However, again it is not possible to place any numerical estimates of the magnitude of this additional layer of uncertainty, as it is situation-specific. For a livestock installation the additional layer of uncertainty might be expected to be small. For other, very different, odours or installations the leap of faith is wider and the additional uncertainty may be much larger. Practitioners must make themselves aware of the possible importance and form a qualitative view on the significance of this component of uncertainty.

On the positive side, the level of annoyance measured by the survey in the New Zealand Technical Report (Ministry for the Environment 2002) was found to be consistent with the odour dose-community-response curves reported by Miedema (1992). The dose-response curves, although developed for other industries and using a Dutch community response, appeared to be valid for pulp mill odours in New Zealand.

In terms of the future outlook for using Indicative Odour Exposure Standards compared to deriving a modelling guideline from a bespoke dose-response study, the Draft H4 guidance does state that the above benchmarks are indicative standards and that UK dose-effect studies are planned. It also states elsewhere in the document that *“the only realistic way of estimating the actual level of annoyance in a particular community resulting from exposure is by carrying out dose-response studies locally”*. However, Draft H4 appears much less explicit than the New Zealand guidance in highlighting the “interim” nature of these generic-type odour guidelines and that they should ideally be superseded by industry-specific guidelines developed from bespoke dose-response studies. It is possible that some dose-response studies will be performed around waste management facilities as part of a study into defining loss of amenity through odour carried out as part of Defra’s Waste Research R&D programme*. There is also a possibility of UK Water Industry Research (UKWIR) coordinating some studies around wastewater treatment plants to support the water industry in meeting the Defra *Code of Practice on Odour Nuisance from Sewage Treatment Works* (Defra 2006).

* Details at <http://www.defra.gov.uk/environment/waste/wip/research/index.htm>

8.4 Correlation with nuisance and complaints

Although the extension of the Draft H4 methodology to predict nuisance or compliance is beyond the scope of this literature review, a brief overview has been given.

The legal use of nuisance has preceded the technical definition of nuisance described here, which has only relatively recently been put forward and generally accepted. The EPA 1990 contains no technical definitions of nuisance, such as maximum concentrations, frequencies or durations of odour in air. This would need to be addressed, together with a nuisance measurement methodology, before any estimate of the uncertainty in the correlation with odour exposure could be made.

Complaints are more usually measured directly by monitoring, rather than prediction. However, dose-response studies using complaints as the response measurand have been carried out in New Zealand and Australia, but using different models to those in common use in the UK and using different percentiles to describe exposure.

It would be safe to assume that the uncertainties in correlating predicted exposure with either nuisance or complaints levels would be considerably higher than for annoyance, due to the additional factors involved.

On the basis of the findings of this study, some of the uncertainties associated with the various steps in a Draft H4 odour assessment may be ranked relative to one another. This is shown in Table 8.1, and is necessarily a simplification.

Table 8.1 Example of Ranking Component Uncertainties

Phase	I	II	III
	Estimation of odour release value	Modelling odour exposure	Correlation with annoyance
Low uncertainty	Frequent DDO on existing source	Flat terrain	Bespoke, UK, industry specific dose-effect relationship
	Single point source	No buildings	Dispersion model same as that used to establish dose-response relationship
	Single, conservative odorant	Choice of met data year and site in flat terrain	
	Continuous, non-time varying emissions	1-hour mean, 'ensemble' modelling	
	Controlled emissions	Defining receptors	
	Conversion to $ou\ s^{-1}$ using ODT for odorant with reliable ODT	On-site met station	
	Many replicate analyses on DDO samples	More than one dispersion n model	
	Application of MCERTS, Environment Agency M1 and M2 guidance, BS EN 13725	Single point source	
		Definition of source velocity, temperature and volume flow – measurements complying with BS EN 13725 and Environment Agency M1 guidance	
Medium uncertainty	Multiple, point sources	Single building	Industry-specific, non-UK dose-effect relationship
	Frequent DDO on similar plant elsewhere	Choice of met site in complex terrain	IOES bands of unpleasantness
	Intermittent DDO on existing source	Choice of new-generation model	Correlation against annoyance
	Batch processes – cycles known	98 th percentile modelling	'Type 2' numerical benchmarks

	Surrogate well correlated with total odour	Multiple point sources	Tolerance factors depending on receiving population
	Few replicate analyses on DDO samples	Coastal effects	
High Uncertainty	Fugitive/diffuse sources	Complex terrain	Dutch livestock dose-effect relationship
	Area/volume sources	Multiple buildings	Correlation against complaints
	Conversion to ou s^{-1} using ODT for odorant with unreliable ODT	Use of fluctuations modelling	Background odours
	Non-conservative odorant/mixture of odorants	Peak-to-mean ratios	FIDOL factors
	Time-varying emissions	99 th - 100 th percentile modelling	'Type 1' deterministic approach
	Estimation of odour release rate using Odour Potential	Area/near-ground/volume sources	Dispersion model different to that used to establish dose-response relationship
	Published Emission Factors	Horizontal releases	
	Source strength dependent on ambient weather conditions/uncontrolled	Capped/obstructed releases	
	Mass balance calculations	Single model	
	Surrogate poorly correlated with total odour	Plume deposition, washout, chemistry	
	Abatement plant efficiency		

9 Recommendations

9.1 Determination of Source Strength

The use of a specific surrogate compound, followed by conversion to $\text{ou}_E \text{m}^{-3}$ using the Odour Detection Threshold (ODT), adds a layer of uncertainty. The uncertainty in this conversion can be the dominant factor affecting the uncertainty of the release value that goes into the atmospheric dispersion model. It is recommended that the Environment Agency makes clear in its guidance that it prefers odour release rates to be obtained directly from $\text{ou}_E \text{m}^{-3}$, and that it does not favour the use of a specific surrogate compound followed by conversion to $\text{ou}_E \text{m}^{-3}$ using the ODT.

For the estimation of the source emission rate, the obvious route for minimising 'human error' would be to ensure all such odour measurements for regulatory purposes are carried out under the MCERTS scheme.

9.2 Odour emission factors

There is a need for an openly available, centralised database for odour emission factors, or at least figures on which uncertainty has been quantified. Currently, commercial organisations have compiled their own databases, which are not widely available. Sparse information is available from disparate sources (e.g. conference publications for agricultural odours, or the waste water industry). A centralised database should be regularly reviewed, upgraded and updated, operator/industrial users of the database should be invited to verify the emission factors.

9.3 Dispersion modelling

UK dose-effect relationships should be established using the latest versions of ADMS *and* AERMOD PRIME, in an effort to move away from benchmarks which were established using dispersion models which are no longer the most up to date with respect to treatment of boundary layer parameters, and which applicants are unlikely to use as part of the odour impact assessment. Such modelling will provide a more robust underpinning for the Draft H4 benchmarks, in their current form or if/once they are revised on the basis of modelling using more recent models and versions thereof.

A model is only 'better' than another for predicting annoyance level if it has been demonstrated to provide a better fit to the regression line in actual dose-effect data for the local situation. In the absence of any robust UK dose-effect survey, model choice based on best-fit of this regression is not possible.

The results of different, new-generation models can vary by a factor of 8, for the calculation of high percentiles. Therefore, it is recommended that more than one dispersion model be used for all assessments. This is particularly important

when buildings and/or area sources are included in the modelling Examination of the range of results would provide a sensitivity analysis of the model algorithms, and provide greater confidence in any regulatory decision.

9.4 Model inter-comparison

Discussions with AQMAU have led Bureau Veritas to understand that the Environment Agency has carried out tests on ADMS-3.3, and comparison against ADMS3.2. The findings of this study, when available, could prove useful for the review of uncertainties in dispersion modelling. In particular, there is concern over the way in which ADMS3.2 and earlier versions calculated the high percentiles, which are relevant for odour assessments.

9.5 Dutch livestock modelling

The original LTDF dispersion modelling data which underpinned setting of the odour standard in the Netherlands has not been available or traceable by UK interested parties (Bull 2004). This continues to be the case.

For the purposes of assessing the validity of the Draft H4 Indicative Odour Exposure Standards, it is important to identify how typical LTDF is of other 'old-generation' models. It would be particularly useful to be able to identify any similarity between LTDF and ISC in their respective predictions of the 98th percentile concentrations. ISC has been subject to most inter-comparison against 'new-generation' models. Technical details on LTDF could not be obtained during the course of Project 3.

9.6 Fluctuations

Dose-effect studies using dispersion models with facilities for estimating fluctuations could be encouraged. Fluctuation modelling provides an opportunity to progress the understanding and development of 'Type 1' numerical benchmarks. This would add to the field validation for fluctuations models. The extent of field validation available for fluctuation modelling, even for conservative pollutants without the complications of odorant FIDOL characteristics, is currently very low. Therefore, it is not yet advisable to set numerical benchmarks based on modelling using fluctuations or peak-to-mean ratios.

Dispersion models are currently in practical use only for predicting ensemble mean (typically hourly mean) concentrations. Fluctuation modelling is not yet adequately validated. As long as this remains the case, the 'Type 2' approach for odour assessment set out in Draft H4 (hourly mean modelling compared against an empirical benchmark) must remain the only feasible option.

9.7 UK epidemiological dose-effect survey

The only realistic way of estimating the actual level of annoyance in a particular community resulting from exposure is by carrying out dose-effect studies locally. Such a study links the exposure (determined by mathematical modelling of emissions from the installation) to the level of annoyance (which is determined by a standardised questionnaire). The relationship between calculated odour exposure and percentage of people annoyed as measured by survey has been experimentally confirmed in the Netherlands, but not in the UK. UK-specific dose-effect studies should be a priority area for research, and have already been identified in such in Environment Agency Report P4-095/TR (Environment Agency 2002b). An application for an existing process could be obliged to carry out a dose-effect survey, to assist in establishing a database. In the Netherlands, the annoyance caused by odour is investigated by systematic year-on-year community surveys (which disguise the purpose to avoid bias). Consideration should be given to establishing surveys or at least representative pilot studies of this kind in the UK

Draft H4 allows PPC applicants to derive industry-specific dose-response relationships between annoyance and modelled 98th percentile odour concentration, as an alternative to using the Draft H4 Indicative Odour Exposure Standards. As yet, the Environment Agency has received no application which has used such a bespoke dose-response relationship. It is likely that a stricter obligation may need to be put on certain operators to establish bespoke relationships for their industry sector.

It is possible that some dose-response studies will be performed around waste management facilities as part of a study into defining loss of amenity through odour carried out as part of Defra's waste research R&D programme.

9.8 Is detailed modelling appropriate?

There is a need to recognise that the uncertainties associated with modelling some types of odorous release (e.g. diffuse/fugitive area sources, non-vertical vents) are very large. In such cases, the use of dispersion modelling as an assessment tool may not be an appropriate use of resources.

It is important that dispersion modelling is not misused. Consideration could be given to incorporating the following guidance into Draft H4.

- Only use odour dispersion modelling for new activities where the predominant odour effect is due to normal process discharges that are continuous or semi-continuous and reliable odour emission rates are available.
- Do not use odour dispersion modelling to try to prove the absence of an adverse effect when community data can be collected, or are available to demonstrate the current level of effect.

- Do not use dynamic dilution olfactometry (DDO) measurement and dispersion modelling to investigate potential acute effects of odour discharge.
- Give careful consideration to the validity of modelling low or area sources in the presence of buildings.
- Give careful consideration to the validity of modelling non-vertical or obstructed releases.
- Give careful consideration to the validity of modelling in the presence of terrain.

9.9 Information required by Draft H4

Draft H4 states that the level of detail supplied should be in keeping with the risk of causing odour-related annoyance at sensitive receptors. More details than those currently stipulated could be requested in the application for an PPC permit. This could include the following, to help to indicate the potential uncertainty:

- Dispersion modelling using more than one model;
- Applicant to assign range of uncertainty to the values of odour emission rate;
- Applicant to fully justify lack of actual source strength measurement for existing process, if applicable. A balance needs to be struck between cost/effort versus benefit/risk;
- For existing process with an existing odour problem, applicant to present findings of a dose-effect survey.

These requirements would need to be set out in the PPC Sector Guidance Notes. However, odour-related issues can be very installation-specific, and the operator may wish to justify why some of this information is not appropriate for a particular site.

ADMFC has provided detailed guidance on best practice for dispersion modelling. Most of the guidelines are applicable to odour modelling. It would be useful if these guidelines were incorporated into Draft H4, giving a clearer steer towards the required level of transparency and rigour in odour assessments. Also, some uniformity in the way that model sensitivity and uncertainty are expressed could be imposed.

9.10 A Checklist for Inspectors

Although beyond the scope of his study, the findings of this Project 3 report, and in particular the information contained in Figures 4.1, 4.2, 5.1, 5.2, 6.1 and 6.2, and Table 8.1, could be used to construct a checklist for Inspectors, to assist with the review of an odour assessment.

Abbreviations and acronyms

ADMLC	Atmospheric Dispersion Modelling Liaison Committee
ADMS	Atmospheric Dispersion Modelling System
AERMOD	United States Environmental Protection Agency Regulatory Dispersion Model
AQMAU	Air Quality Modelling and Assessment Unit
AQS	Air Quality Strategy
BAT	Best Available Technique
BS	British Standard
BV	Bureau Veritas
CEN	Comité Européen de Normalisation/European Committee for Standardisation
CEM	continuous emissions monitor
CERC	Cambridge Environmental Research Consultants
CFD	computational fluid dynamics
CIWEM	Chartered Institution of Water and Environmental Management
CRM	certified reference material
DDO	dynamic dilution olfactometry
Defra	Department for Environment, Food and Rural Affairs
Draft H4	<i>'Horizontal guidance for odour'</i> , Environment Agency IPPC H4 Integrated Pollution Prevention and Control, October 2002
EFD	Emission Factors Database
EIA	Environmental Impact Assessment
ELV	Emission Limit Value
EPA	Environmental Protection Act or Environmental Protection Agency

EROM	European Reference Odour Mass
FIDOL	frequency, intensity, duration, offensiveness, location
FPD	flame photometric detector
GIS	Geographical Information Systems
IOES	Indicative Odour Exposure Standard
IPC	Integrated Pollution and Control
ISC/ISCST	Industrial Source Complex/Industrial Source Complex Short Term (dispersion model)
ISO	International Organisation for Standardisation
LTDF	Lange Termijn Frequentie Distributie (Dutch regulatory model)
MCERTS	Monitoring Certification Scheme
NRPB	National Radiological Protection Board
OCI	odour concentration–intensity (relationship)
ODT	odour detection threshold
OP	Odour Potential
ou _E m ⁻³	European odour units per cubic metre of air
NH ₃	ammonia
PDF	probability distribution function
PPC	Pollution Prevention and Control
ppm	parts per million
PT	proficiency testing
QA/QC	quality assurance/quality control
R&D	research and development
RMetS	Royal Meteorological Society
STA	Source Testing Association

STOP	Sewage Treatment Odour Prediction model
U	uncertainty
UKWIR	UK Water Industry Research
UV	ultraviolet
VDI	Verein Deutscher Ingenieure (standards)
VOC	volatile organic compound
WASP	Workplace Analysis Scheme for Proficiency

Appendix A – Assessment of Uncertainty of Stack Emissions

A.1 The general principles in assessing uncertainty

Though there are a number of different methods of assessing uncertainty, the general principles are as follows:

- i Specification* - define what is being measured and the parameters on which it depends;
- ii Identify the uncertainty sources* - for each parameter in the relationship, list the possible sources of uncertainty, e.g. sampling, instrument bias, reagent purity, environmental conditions;
- iii Quantify the components* - measure or estimate, then calculate total uncertainty using accepted rules.

i and ii Specification and identification of the sources of uncertainty

The overall uncertainty that should be considered is that of the *whole* measurement*. The measurement is made up of a sampling stage and an analytical stage. The sampling and analysis equipment will be an important, but not the only, component of this. In this conceptual model, measurements using direct-reading instruments (e.g. for hydrogen sulphide) should be treated no differently to measurements where a sample is collected discretely and then analysed later (e.g. by dynamic dilution olfactometry, DDO) on the laboratory bench. Both have sampling and analysis stages: for example, a chemiluminescence analyser samples the stack gas through a heated line, conditions the gas and then presents it to the reaction cell where the analysis takes place.

It is important not to use restrictive definitions of sampling and analysis: in this context they mean the collection of the measured property and its quantification, respectively. Sampling can include, for example, the placing of a thermocouple and pitot at a particular point in a stack to obtain the information to calculate the volume flow (necessary to get the mass odour emission rate from the concentration). The analysis stage in this example is the instrumental conversion of the physical properties to electronic readouts of temperature and pressure.

* This will not necessarily be the case for organisations whose involvement is restricted to one portion of the measurement, e.g. an analytical laboratory providing the analysis results, or an instrument manufacturer providing the equipment. These organisations will typically quote uncertainties relating only to their own portion of the overall stack emission measurement.

Both the sampling and the analysis have many component uncertainties that go to make up the overall uncertainty. Some examples are shown in Figure A.1. This list is not exhaustive, and careful thought needs to be given to the potential sources of uncertainty in the method at each of these stages. This conceptual model is not specific to one class of technique and may be applied to odour measurements, using continuous emissions monitors (CEMs) (e.g. for H₂S emissions) or periodic monitoring (e.g. lung sampling followed by DDO), or using instrumental or classical techniques. Here again, it is helpful to avoid interpreting terms too restrictively. A number of examples follow.

- Losses during transport and conditioning of the sample would apply equally to errors from condensation and adsorption in the heated line of a continuous analyser, and to losses due to degradation and adsorption on container walls during transport of a bag sample solution from a periodic stack test to a laboratory for later DDO analysis.
- Sampling collection medium efficiency would apply equally to the collection efficiency of a sorbent solution in an impinger, and the collection efficiency of a solid-sorbent resin such as Tenax.
- For analysis, calibration uncertainties could include the calibration error for an instrumental analyser (including the tolerance on the traceable calibration gas standard), and for DDO could include the uncertainty in the standard reference odour due to the tolerance of the reagent purity.
- Volume errors for a classical wet analysis would include the tolerances on the volumetric glassware used (e.g. measuring cylinders, pipettes, burettes, etc.), whilst for instrumental techniques would include tolerances on reaction-cell volumes, sample-loop volumes, etc.

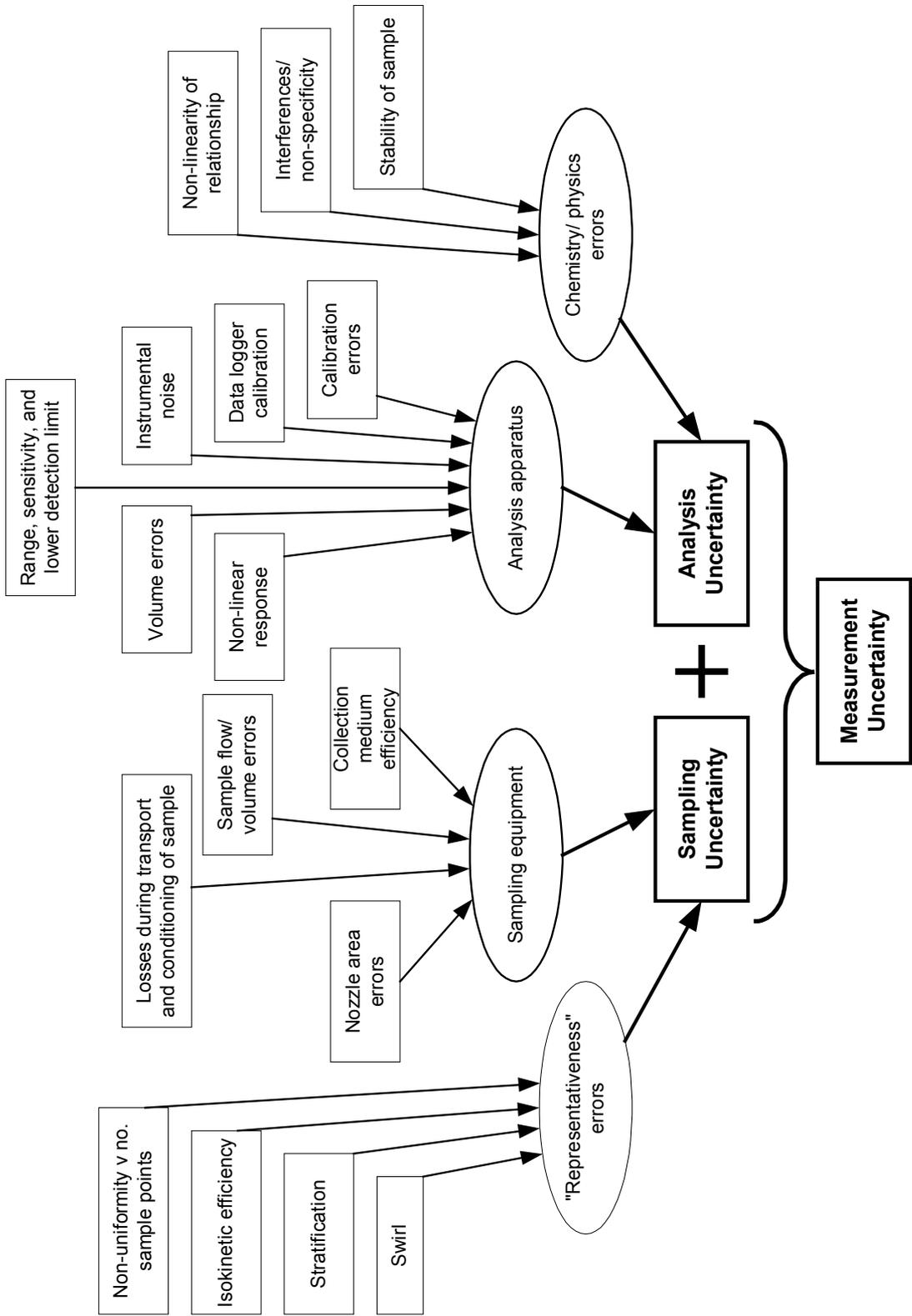


Figure A.1. Examples of component uncertainties contributing to the overall uncertainty

iii Quantifying the uncertainties

There are four main approaches to quantifying uncertainties, summarised in Table A.1. Different approaches are appropriate to different situations. Furthermore, some of these approaches give better estimates of uncertainty than others: the best provide information on the combined effects of the reproducibility of the measurements and the bias (the difference between a measured value and the “true” value); others give a reasonable estimate of the uncertainty arising from random effects (scatter) but do not address systematic, or bias, uncertainties.

Table A.1 The four main approaches to quantifying uncertainties in source emissions measurements

Approach	Advantages	Disadvantages
A. Repeat measurements on reference materials	“Real” values	Reference materials not always available, e.g. particulates suspended in a gas stream. If carried out in the lab, difficult to include all the variables that may have an influence on site.
B. Experimental work, e.g. repeatability experiments, paired comparisons, and ring tests	“Real” values	Gives good estimate of precision, repeatability, etc. but often fails to include bias. Can be expensive, e.g. CEN ring tests up to €100,000 per test with multiple teams
C. Estimations based on previous results/data, e.g. instrument specifications, calibration data, PT schemes	Can “unpick” the uncertainty budget and rebuild if there are changes, e.g. deviations, new equipment.	Can appear rather abstract.
D. Estimations based on considered judgement	This is usually considered to be the method of last resort	

A.2 How to Assess Uncertainty in Practice

With the advent of MCERTS, organisations carrying out regulatory monitoring for Integrated Pollution Control (IPC) or PPC purposes will be using measurement methods specified in Environment Agency Technical Guidance Note M2 (Environment Agency 2004). For most - but not all – determinands, published standard methods (e.g. CEN, ISO or BS) are specified. Where a determinand has no current published standard, M2 will nevertheless list the measurement techniques that can be *used*; a test house will then need to document an in-house method based on one of these techniques. Even when a standard method has been published, it will not necessarily quote an uncertainty value. Furthermore, it is not always possible to carry out a measurement complying with every aspect of the standard. This leads to three main scenarios, summarised in Table A.2.

Table A.2 Actions needed to assess uncertainty

Scenario	Action needed
1) Measurement to a published standard method that quotes an uncertainty estimate. Carried out in strict compliance with standard, with no deviations.	Quote the uncertainty given in the standard.
2) Measurement method has no uncertainty estimate quoted, either because it is an older-type standard, or because the test house has had to document its own in-house method based on the techniques recommended in M2	Estimate the uncertainty for this method using one of the four approaches discussed previously. Quote this uncertainty value if the measurement is carried out in strict compliance with the standard or documented method.
3) Measurement deviates or is not in strict compliance with the published standard or documented in-house method.	Estimate a revised uncertainty, taking into account the deviations.

In Scenario 1, a published standard measurement method is used that quotes an uncertainty. However, this uncertainty will only be achieved if the exacting requirements of the standard method are complied with in full. In practice, this can be difficult because of, for example, poor positioning of the sample plane allowing only some of the ports to be accessed or leading to stratification. There may also be other deviations from the exact procedure stated in the method. In these circumstances a bespoke estimation of the uncertainty for the test must be calculated. Obviously, if the measurement method used does not quote an uncertainty at all, this will need to be estimated using one of the approaches described above.

Caution should be exercised when using quoted estimates of uncertainty from published methods, instrument specifications, etc. There has been considerable confusion among non-specialists of terms such as accuracy, precision, reproducibility, uncertainty, etc. and in older documents it is not uncommon to find these terms used interchangeably or incorrectly. Even when the term uncertainty is used correctly, it must be remembered that there are different approaches to its estimation. This means that what is quoted simply as “uncertainty” in a standard or technical specification may require a closer look to reveal just what aspect of uncertainty is being expressed. For example, BS EN 1911 for hydrogen chloride quotes “internal uncertainty” and “external uncertainty”, which correspond to reproducibility and repeatability, respectively. These parameters do not include all the bias uncertainties in the classical definition of overall uncertainty.

In the following sections, the approach of building up an uncertainty budget is considered. This bottom-up approach has been used previously in STA guidance for estimating bespoke uncertainties for some older, published measurement methods, e.g. BS3405 for particulate.

A.3 The Uncertainty Budget Approach

Overview

As with all the approaches, the general principles that apply are:

- i. specification of the component parts of the measurement;
- ii. identify the sources of uncertainty for these components; and
- iii. quantify uncertainty components and combine them.

In the first stage, it is necessary to think about all the significant steps in the measurement, including both sampling and analysis. The second stage involves looking critically at these steps and listing the sources of uncertainty.

This may result in a large number of component uncertainties. Because the third step involves quantifying these, it is useful to focus on a shortlist of those making the most significant contributions. By convention, those components expected to contribute less than a third of the largest need not be evaluated in detail.

Quantifying the component uncertainties

This is perhaps the most challenging step. Information on the magnitude of the component uncertainties may often be found in the following sources.

- *Published standards, especially the most modern ones* – often limit specifications are placed on some of the measurement components, e.g. sample volume must be measured to within $\pm x\%$, nozzle area must be within $\pm y\%$ of the nominal value;
- *Manufacturers' specifications* – the technical specifications for an instrument or piece of apparatus will often give valuable information on the component uncertainties. For example, an infrared analyser spec may state that temperature drift is $\pm x\%$ between 0 and 50 °C. A wet-chemistry example would be a 500 ml measuring cylinder used to measure the volume of sorbent solution from the impingers. If the glassware conforms to BS604:1982, the graduations will be every 5 ml and the maximum permitted error in the graduations will be 5 ml. The error in the cylinder plus the error in reading will therefore be up to 7.5 ml.
- *Calibration certificates* – such data can be useful in a number of ways: calibration certificates, such as those provided for calibration of balances, may themselves quote the degree of uncertainty for the calibration. Once a number of successive calibrations have been carried out, the drift can be estimated. In other applications, a tolerance is quoted. For example, a calibration gas may state a nominal concentration $\pm 5\%$.
- *QA/QC data* – much of the data produced as part of a monitoring organisations ongoing quality system can be used in the uncertainty budget. For instance, daily drift-checks on balances, daily checks on nozzle diameters and monthly checks on thermocouples and manometers may provide better estimates of their component uncertainties than simply taking as default values the limit specifications in the published standard. Similarly, the pre- and post-sampling span and zero checks that should be made on

direct-reading monitoring instruments will provide valuable “real” information on the drift that is found under true site conditions. Other valuable information will come from inter-laboratory proficiency testing (PT) schemes. Some, such as the various schemes for chemical analysis, e.g. Workplace Analysis Scheme for Proficiency (WASP), give information on the uncertainty of a particular part of the measurement (in this example, the analysis) that can be used in the overall uncertainty budget. It should be pointed out that the Stabs gas analysis PT scheme looks at the variability between different test houses in making the whole gas concentration measurement: accordingly, this information is better considered as a separate, alternative (Approach B) estimate of the overall uncertainty, rather than as part of an uncertainty budget (Approach C).

As noted earlier, estimates of component uncertainties will be found expressed in many different ways. However, for building up the uncertainty budget it is crucial that consistency is applied. Firstly, all of the component uncertainties must be expressed in a common unit. If necessary they must be converted to this common unit. Once all the component uncertainties have a common unit, they each need to be expressed as the *standard uncertainties*, i.e. one standard deviation. The STA guidance booklet gives further help on this, recommending that the component uncertainties are listed under two categories:

Type A component uncertainties - these are data obtained directly from experiments, observations or measurements and it should be straightforward to express them as the standard deviation; and

Type B component uncertainties - these are the kind of figures often provided in instrument specifications, with $\pm X$ values but no associated confidence limits. In these cases the standard deviation is estimated as $X \div \sqrt{3}$.

Combining the component uncertainties

Once the individual component uncertainties have been listed, short-listed and then quantified, they must be combined to produce an estimate of the *overall* measurement uncertainty. The component standard uncertainties cannot simply be added. Although there are a number of statistical approaches that have been used, the simplified method recommended by Eurochem involves taking the square root of the sum of the squares of the component uncertainties, as described in the STA guidance:

$$u_{\text{combined}} = \sqrt{(u_1^2 + u_2^2 + \dots + u_n^2)}$$

Finally, the combined standard uncertainty is multiplied by the appropriate *coverage factor* to give the *expanded uncertainty*. By convention 95% confidence limits are used, for which the coverage factor is 1.96, usually rounded-off to 2.

Expressing the overall uncertainty

As discussed earlier, it is essential that the component uncertainties were converted to a common unit before they were combined. Often, this will have resulted in an overall uncertainty expressed in units of concentration (e.g. ppm). It is sometimes useful to show the uncertainty in relative terms, expressed as a percentage of a reference concentration. For an instrumental method, this is

often taken as that corresponding to 50% of the full scale reading. So if the instrument is used typically on the 0-1000 ppm range, the overall uncertainty will be expressed as a percentage of 500 ppm.

$$\text{Overall uncertainty (\%)} = (U \times 100) / X_{ref}$$

It should be noted that if the mass emission is required to be calculated from the concentration and the volume flow, then this will have a larger uncertainty. According to Hawksley *et al* (1977), the uncertainty is increased by a factor of $\sqrt{2}$, provided the stack gas flow has been measured according to BS3405.

The STA guidance booklet gives worked examples of uncertainty budgets for instrumental methods, wet-chemistry methods and particulate monitoring methods.

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